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LAKESHORE CAPACITY STUDY

COMMITTEE REPORT

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LAKESHORE CAPACITY STUDY

COMMITTEE REPORT

JULY 1986

Prepared by:
JEAN C. DOWNING, M.A., M.C.I.P.

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Ontario
Ministry of Municipal Affairs
Bernard C. Grandmaître, Minister
Research and Special Projects Branch

The following Lakeshore Capacity Study reports

Committee Report
Land Use
Fisheries
Microbiology
Trophic Status
Wildlife
Integration

are available from:

Ministry of Municipal Affairs
Research and Special Projects Branch
777 Bay Street, 13th Floor
Toronto, Ontario
M5G 2E5

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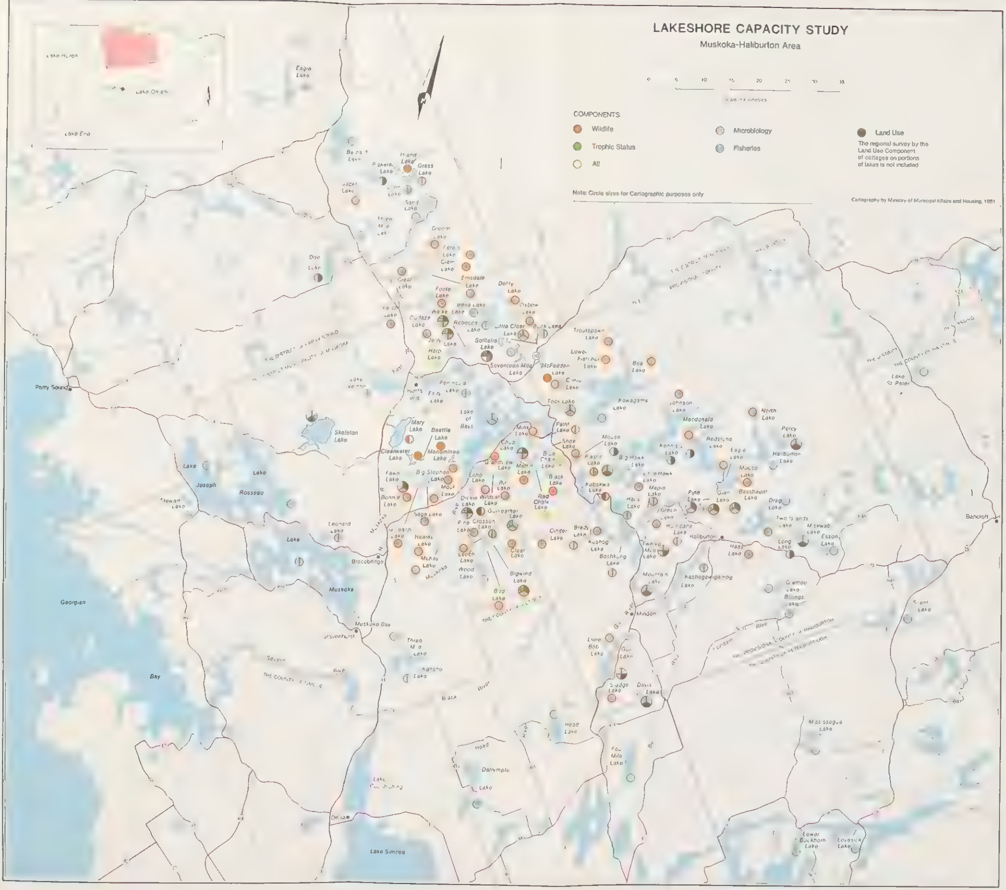
LAKESHORE CAPACITY STUDY
Muskoka-Haliburton Area
COMPONENTS
Wildlife
Trophic Status
All
Microbiology
Fisheries
Land Use
The regional survey by the Land Use Component of cottages on portions of lakes is not included
Note: Circle sizes for Cartographic purposes only
Cartography by Ministry of Municipal Affairs and Housing, 1981
Parry Sound
Skeleton Lake
Joseph
Rosseau
Muskoka
Bracebridge
Muskoka Bay
Georgian
Bay
Lake Simcoe
Orillia
Haliburton
Minden
Bancroft
Mary Lake
Beattie Lake
Clearwater Lake
Menominee Lake
Solitaire Lake
Seventeen Mile Lake
McFadden Lake
Paint Lake
Shoe Lake
Red Chalk Lake
Big Stephen Lake
Blue Chalk Lake
Black Lake
Plastic Lake
Pine Lake
Bird Lake
Bigwind Lake
Davis Lake
Dalrymple Lake
Head Lake
Four Mile Lake
Lower Buckhorn Lake
Lovesick Lake
Mississagua Lake
Black River

LAKESHORE CAPACITY STUDY

STEERING COMMITTEE

Ministry of Municipal Affairs and Housing
M.H. Sinclair Chairman
J.C. Downing
Ministry of the Environment
S.E. Salbach
Ministry of Natural Resources
R.A. Riley

STUDY COORDINATOR

G.C. Teleki
Ministry of Municipal Affairs and Housing
Now with DeLeuw Cather, Canada Ltd. (DeLCan)

FOREWORD

Community planners and other professionals involved in the preparation of planning policies for lakeshore development and in the review of specific subdivision proposals have always found it difficult to determine objectively the impact of development on the natural environment. In response to this challenge, the Lakeshore Capacity Study was undertaken to provide a planning tool to assist in evaluating the effects of cottage development on inland lakes and lakeshores. Central to the task was the need to gain a clearer understanding of the relationship between cottage development and its impacts on selected aspects of the natural environment.

To accomplish these objectives, the Ministry of Municipal Affairs carried out the Lakeshore Capacity Study in cooperation with the Ministry of the Environment and the Ministry of Natural Resources.

The Muskoka-Haliburton area of central Ontario was chosen as the Study Area. The homogeneity of the area, which is part of one physiographic region, reduced the need to account for major natural variations among the lakes and watersheds. In addition, the extent of existing development on the lakes varied, permitting an examination of situations ranging from no development to full development.

The Study involved measuring the source of environmental impact, in terms of the lakeshore cottages and their use, and the impact of cottage development in terms of water quality, fisheries, and wildlife habitat. The methods of prediction derived from the research were linked in a simulation model, which is capable of predicting trends in the values of the key indicators of impact.

As the research, analysis and findings of the Study are documented in a set of seven reports, selective reading may be desirable. Those readers who have a general interest in the work are advised to read this Committee Report first, as it provides an overall summary of the findings. This should be followed by the Integration report, in which the simulation model is described. Readers with more specialized interests will find the details of each component of the Study in the other five reports. In each, the reader can select from the table of contents the most important chapters for his or her purposes.

The end product of the Study, the Ontario Lakeshore Capacity Simulation Model, has several features worth noting. The spatial unit addressed by the model is a single lake and the lakeshore. When the model is applied, the number of unknowns related to the natural environment can be substantially reduced, making it easier for planners or other professionals to weigh the environmental effects of development. The model goes a step further to permit predictions of the impact of cottage development when different management policies are selected.

The scope of the simulation model demands some explanation. In its present form, the model applies to cottage development on inland lakes in the Study Area, where the research was conducted. However, the methods of prediction

can be adapted to other parts of the province, as long as differences in conditions are taken into account.

The purpose of the Study was to measure the environmental impact of cottages. Commercial and industrial uses were excluded deliberately, in order to simplify the difficult task of measuring cottage impact. The flexibility inherent in the simulation model makes it possible to add other types of land use later, if so desired.

Most of the existing cottage development in the Study Area is located in a single tier along the shoreline. For this reason, the simulation model applies to the immediate lakeshore and not to backshore development. Again, the methods of prediction developed for cottages near the lake can be adapted to measure the impact of cottage development in other forms.

The model was designed to measure the physical and chemical impacts of cottages. Accordingly, it does not address other planning concerns, such as social and economic impacts. While these were recognized as essential considerations in decision-making, the specific objective of Phase III of the Lakeshore Capacity Study was to find practical ways of producing better technical data regarding environmental impact.

Now that Phase III of the Study is completed, with the production of the Ontario Lakeshore Capacity Simulation Model (OLCSM), the next step envisaged is to apply the model experimentally within the Ministry. In this setting, model output can be tested in a variety of actual development situations. When this period of experimental use has been concluded and the results assessed, the three participating ministries will be able to determine whether the model should be adapted to other parts of the province and whether it should be made available more widely.

The Ministry of Municipal Affairs considers the OLCSM to be a potential planning tool but recognizes that the technical and organizational implications of its use must be examined. While this is underway, the model will be available for testing as an additional planning tool to supplement the information normally required to evaluate a planning policy or development proposal. However, the model will not be used in the decision-making process, which will still rest on the customary range of planning considerations.

This Committee Report describes the Lakeshore Capacity Study results in terms of their underlying principles and practical applications.

ACKNOWLEDGEMENTS

The material in this report is drawn largely from reports prepared by the managers and senior staff of individual components of the Lakeshore Capacity Study. A conscious effort has been made, within the bounds of a general summary, to convey the essence of their original work.

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PREFACE

This report is addressed to planners and others who are interested in a general overview of the Lakeshore Capacity Study. The focus is on results, with sufficient background provided to make them comprehensible. Models developed by the research teams to predict cottage use and the impact of lakeshore development on the land-lake system are described in terms of their underlying principles and practical capabilities.

A more detailed account of the Ontario Lakeshore Capacity Simulation Model (OLCSM) appears in the Integration report. It describes the submodels and their linkages more fully and includes all the equations.

Those interested in the research design and methods will find a record of the supporting research and results in the other Lakeshore Capacity Study reports: Land Use, Trophic Status, Microbiology, Fisheries and Wildlife. These reports document the data collection procedures, analytical methods, research results and modelling processes, together with the mathematical formulations of functional relationships in the models.

The Summary at the beginning of this Committee Report presents the results of the Lakeshore Capacity Study in terms of their relevance to lakeshore planning and development.

Chapter 1 outlines the objective and scope of the Lakeshore Capacity Study and its management structure. Part A describes the Ontario Lakeshore Capacity Simulation Model (OLCSM) and its potential as a practical planning tool. Part B reviews the research results and the methods which were developed to predict lakeshore cottage use and the impact of lakeshore development on water quality, public health, fisheries and wildlife.

SUMMARY

This Summary presents the results of the Lakeshore Capacity Study in terms of their relevance to lakeshore planning and development. The material is arranged in the same sequence as the report.

INTEGRATION COMPONENT

The Integration Component team constructed a state-of-the-art simulation model to predict the effects of lakeshore cottage development on water quality, fisheries and wildlife, for inland lakes in the Study Area. The adaptive environmental assessment and management (AEAM) approach inherent in the Ontario Lakeshore Capacity Simulation Model (OLCSM) is specifically designed to assist in ecological decision-making in real-world situations.

The OLCSM is technically sound, based as it is on relationships and methods derived from the Lakeshore Capacity Study research. The innovative work of the research component groups, with supplementary knowledge from the scientific literature, is the most advanced in the field.

While the Lakeshore Capacity Study research involved complicated concepts and analyses, the end product of the Study is a relatively simple, practical planning tool. The Ontario Lakeshore Capacity Simulation Model responds to simple commands, operates with readily available data and generates the predictions quickly.

The OLCSM offers several advantages over four separate models. First, explicit linkages between the submodels simulate ecological interactions more realistically, as natural processes operate without regard to the areas of responsibility of specific ministries. For example, water and fish are inseparable ecologically, even though they may be separated for administrative purposes. Second, input data for a particular lake is entered into the computer only once, thus ensuring consistency in the data base for all the predictions. Third, the OLCSM predicts over time the values of indicators that measure the state of the environment. As the values are calculated for each annual time-step, the user will know when any undue stress on the environment is likely to begin and can act accordingly. Fourth, the OLCSM can simulate the effects of different management strategies designed to mitigate the impact of development on the environment.

LAND USE COMPONENT

The models developed by the Land Use Component planners and social scientists, as part of the Lakeshore Capacity Study, are capable of predicting cottage use, for a specific lake, with greater precision than was previously possible. The Accessibility and Cottager models are sensitive to variations in cottage use between lakes and between cottages. Accordingly, more realistic estimates can be obtained by using the models than by assuming maximum use of each cottage.

The value of realistic cottage use predictions is evident in the context of the simulation model. When the predictive methods developed by the individual components are incorporated into the Ontario Lakeshore Capacity Simulation Model, the prediction of cottage use becomes the driving variable. In the Water Quality submodel, for example, estimates of anthropogenic phosphorus input to the lake from septic systems are derived, in part, from the predicted annual user-days. Thus, good predictions of cottage use tend to improve the predictions of environmental impact.

Quite apart from the Ontario Lakeshore Capacity Simulation Model, the Land Use Component models can be used independently to identify lakes where the pressure for cottage conversion is likely to be relatively high. Such technical information could assist planners who are responsible for drafting appropriate official plan policies and zoning regulations with respect to seasonal versus permanent residential uses on specific lakes.

TROPHIC STATUS COMPONENT

Models developed or refined by the Trophic Status Component scientists, as part of the Lakeshore Capacity Study, are capable of predicting the impact of lakeshore cottage development on the trophic status of lakes in the Study Area with greater accuracy than the models available previously.

In predicting the trophic status of a lake, the effects of upstream lakeshore development must be considered, as inflowing streams are one of the sources of phosphorus. The trophic status models can be used for each lake upstream, starting with the headwater lake, in order to calculate the phosphorus load in streams flowing into the lake where new development is proposed.

The Ministry of the Environment is responsible for surface water quality management. For several years, as part of the Ministry's water management program, the existing trophic status models have been used to assist in evaluating the effects of lakeshore development. The new Trophic Status Component models, based on the best scientific knowledge available, offer a more refined tool.

MICROBIOLOGY COMPONENT

Scientists in the Microbiology Component developed, as part of the Lakeshore Capacity Study, a better understanding of the relationships between high densities of swimmers, water quality and the risk of ear infection. When this was combined with the Land Use Component findings regarding cottage use and swimming activity, it was evident that extensive cottage development on the lakeshore could generate a higher risk of ear infection among swimmers.

The water quality criterion recommended by the researchers provides a specific benchmark for evaluating the risk of ear infection. The Ministry of the Environment is now using *Pseudomonas aeruginosa* as an indicator of the health of recreational waters, so their responses to the Ministry of Municipal Affairs regarding a proposed subdivision reflect this criterion.

The biologists' recommendations for beach location provide practical guidelines for minimizing waterborne concentrations of the pathogen *Pseudomonas aeruginosa*. The application of such guidelines could reduce the risk of swimmers contracting the ear infection *otitis externa*.

Finally, the recommendation to design beaches to encourage low densities of swimmers leads planners towards health-conscious site planning on the shores of inland lakes. Given the knowledge that high human loading of a beach can increase the risk of ear infection for swimmers, the lakeshore can be planned and developed to minimize risks. If the proposed cottage development is likely to generate a large number of swimmers or if the beach is also to be used by non-cottage swimmers, more than one beach may be appropriate. Alternatively, access to a large beach may be designed to distribute the swimmers more evenly along the shoreline. A method of calculating swimmer density and relating it to the risk of ear infection is included in the Ontario Lakeshore Capacity Simulation Model.

FISHERIES COMPONENT

The net productivity model developed by the Fisheries Component biologists, as part of the Lakeshore Capacity Study, is designed to estimate the fish catch, or fish yield, for a specific lake. When the predicted yield is compared with the estimated maximum sustained yield, fisheries biologists are able to judge whether the fishery is likely to be overexploited.

The net productivity model can be used to predict the yield not only for the fish community but also for one species. This is accomplished by using the estimated MSY (maximum sustained yield) value for the species in question, for example, lake trout. This model capability can assist fisheries biologists in evaluating the impact of lakeshore cottages on lake trout lakes.

The Ministry of Natural Resources is responsible for the management of commercial and sport fisheries in Ontario. As part of their fisheries management program, the ministry has used the fish yield estimated from the existing morphoedaphic index model as one of the criteria for setting catch quotas for some of Ontario's fisheries. The net productivity model, developed as part of the Lakeshore Capacity Study, offers a more refined tool for predicting fish catch for lakes with existing or proposed lakeshore cottage development.

WILDLIFE COMPONENT

Biologists in the Wildlife Component developed, as part of the Lakeshore Capacity Study, quantitative methods for predicting the impact on wildlife communities of residential development on the shores of inland lakes in the Study Area. Their innovative research provides new knowledge with respect to wildlife habitat needs.

The challenge for planners is to recognize the opportunities for coexistence of cottages and wildlife. Fortunately, some of the land requirements for wildlife habitat differ from those for cottages. Streams and wetlands, for example, provide ideal habitat for several wildlife species but poor sites for cottages.

The Ministry of Natural Resources has a specific mandate under the Endangered Species Act to protect endangered, rare and threatened species of wildlife. While none of these occur in the Study Area, studies of the red-shouldered hawk suggest that its nesting habitat is being reduced and that the population is declining.

The deer population is also a concern of the Ministry of Natural Resources. From a land use standpoint, this involves preserving deer concentration areas which are important to winter survival. Regional staff of the Ministry of Natural

Resources can provide information regarding the locations of deer yards, so these can be taken into account in planning for cottage development.

The wildlife recommendations provide guidelines for those who wish to improve lakeshore planning and development. By taking the needs of the wildlife community into account at the planning stage, developers may be able to enhance the potential of their cottage lots and thereby enlarge their share of the market.

The results of the Wildlife Component research can also benefit cottagers. With greater knowledge of the types of habitat needed by various wildlife species, cottagers will be able to avoid destructive action. Those who value the wildlife near their cottages may limit the removal of vegetation on their lots. Others may make a conscious effort to avoid disturbing nesting sites. While the actions of one cottage household are relatively small in scale, the cumulative effects could be considerable.

In short, the wildlife research results offer a wide range of pertinent knowledge which concerns cottagers and developers as well as biologists and planners.

1. INTRODUCTION

1.1 PROBLEM

For many years, community planners in Ontario have had difficulty evaluating subdivision proposals for cottage development on the shores of inland lakes. The problem stemmed from the absence of technical methods with which to determine objectively the impact of cottage development on the lake and surrounding land.

Similar problems were encountered in determining what policies should be included in regional and local official plans. Even more specific aspects of lakeshore planning, such as zoning standards and site planning, were hampered by the lack of relevant technical knowledge.

Without a recognized method of predicting impact, conflicting claims regarding the ecological repercussions of lakeshore cottages tend to generate confrontation. While development proponents may minimize the adverse effects, cottage owners often foresee dire consequences. Any reconciliation of such extreme positions is unlikely, as long as crucial gaps remain in scientific knowledge of the ecological relationships inherent in lake-watershed systems.

With a method of predicting impact prior to actual development, damage to the environment could be prevented by limiting the number of cottages to the carrying capacity of the lake. Such an approach would be more economical over the long term, as it would avoid the need for costly remedial action in the future.

1.2 OBJECTIVE

The objective of the Lakeshore Capacity Study Phase III was to develop a systematic method of predicting the impact of lakeshore cottages on selected aspects of the inland lake environment. It was clear from the initial work in Phases I and II that an in-depth examination of certain specific lake features, such as water quality and fisheries, could be expected to produce practical results. Further, the land elements of the land-lake system could be taken into account through wildlife-habitat investigations. Related to each of these was the need to find more effective ways of quantifying cottage development on the lakeshore, to enable identification of consistent relationships between cottages and water quality, fisheries and wildlife.

A practical necessity was that the methods, or models, must be able to produce projections without delay, so they could be used to expedite planning processes. To accomplish this, the input data would need to be assembled quickly. Thus, the variables and parameters selected as predictors would have to be restricted to those for which values would be readily available.

In view of the limits on current scientific understanding of lake-watershed systems, the model to be developed in Phase III was not viewed as the ultimate solution. Rather, a more realistic expectation was that it would be sound enough to merit practical application and flexible enough to allow future refinement.

Finally, it was clear that no model could solve all lakeshore planning problems. Nevertheless, by measuring the probable impacts of cottage development, the model projections could substantially reduce the number of unknowns with respect to environmental impact, making it easier for a planner to weigh the environmental factors along with other considerations. In this way, the model would be able to assist in lakeshore planning at the provincial, regional and local levels.

1.3 SCOPE

The Study Area selected was the Muskoka-Haliburton area of central Ontario. As this area is within one physiographic region on the Precambrian Shield, the soils and plant communities are similar throughout the area, thus reducing the need to account for major natural variations among the lakes and watersheds. With respect to cottages, the extent of existing lakeshore development varies, permitting examination of situations ranging from no development to full development.

The research was concerned with the physical and chemical impact of lakeshore cottages on the land-lake system, leaving other types of impact and land use to be investigated at some future time.

1.4 MANAGEMENT STRUCTURE

The Lakeshore Capacity Study research work was carried out within three ministries: Municipal Affairs and Housing, the Environment, and Natural Resources.

STEERING COMMITTEE

The Study was initiated, funded and managed by the Ministry of Municipal Affairs and Housing. In Phase III, a Steering Committee was established, with representatives from the participating ministries. The role of the Committee was to ensure that the study objectives were clearly understood, the approved research program was followed and the resources were properly allocated.

Table 1. Research management

Ministry	Component	Manager
Municipal Affairs and Housing	Integration	Géza C. Teleki*
	Land Use	Jean C. Downing
Environment	Trophic Status	P.J. Dillon
	Microbiology	C.A. Burger
Natural Resources	Fisheries:	
	Net Productivity	A.M. McCombie
	Littoral Zone	J.M. Harker**
	Wildlife	D.L. Euler

* Now with DeLeuw Cather, Canada Ltd. (DeLCan)
** Now with M.M. Dillon Ltd.

RESEARCH COMPONENTS

The research was carried out within several components in the three ministries. Managers of the various components were senior staff with knowledge and experience in research and in appropriate disciplines, including limnology, biology, microbiology and planning. The research management structure is set out in Table 1.

PEER REVIEW PROCESS

The Steering Committee introduced a peer review process to ensure that the Lakeshore Capacity Study research was technically valid, current and complete (Figure 1). Eminent scientists and planners from the private sector and from public agencies outside the Ontario public service were invited to act as peer reviewers (Appendix A). They examined the research design, methods and reports of individual components and provided constructive comments. This peer review process proved to be a valuable adjunct to the work of the research teams.

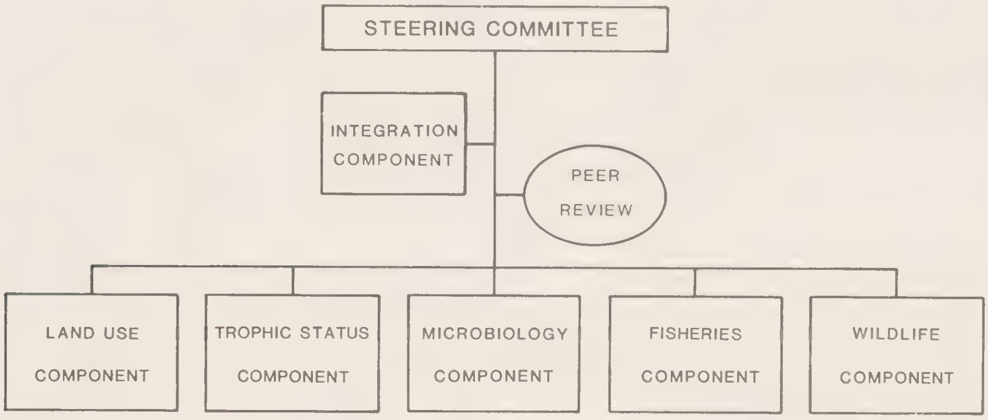

Figure 1. Lakeshore Capacity Study management structure

1.5 ACCOMPLISHMENTS

In each component, the research proceeded independently, within the framework of the Lakeshore Capacity Study as a whole. In some components, such as Trophic Status and Fisheries, the scientists were able to build on existing models; in others, such as Wildlife and Land Use, the researchers had to do pioneering work. Despite substantial differences in the nature of the research and the results, each component produced original, technically sound results, with practical implications for lakeshore planning.

Methods of predicting cottage use and the impact of lakeshore cottages on water quality, fisheries and wildlife were developed and incorporated into the Ontario Lakeshore Capacity Simulation Model (OLCSM), which is also capable of testing the effects of different management strategies designed to mitigate environmental impact. Thus, the objective of the Lakeshore Capacity Study Phase III has been achieved.

PART A. ONTARIO LAKESHORE CAPACITY SIMULATION MODEL

2. INTEGRATION COMPONENT

3. POTENTIAL APPLICATION OF THE ONTARIO LAKESHORE CAPACITY SIMULATION MODEL

2. INTEGRATION COMPONENT

The goal of the Integration Component was to combine the methods of prediction developed by the other Lakeshore Capacity Study components into a single predictive tool which would be capable of quantifying the effects of lakeshore cottage development on water quality, fisheries and wildlife.*

From a technical standpoint, the integrated model had to be able to produce sound information regarding selected indicators of the environment. Further, it must be flexible enough to allow revision and improvement based on new scientific knowledge and experience.

From a practical standpoint, the model had to be reasonably simple for users to apply. In addition, it must be able to generate predictions quickly and present them clearly.

2.1 EXISTING MODEL

An adaptive approach to environmental impact assessment and management was developed in the 1970's at the Institute of Resource Ecology, University of British Columbia. The approach is described in a book edited by C.S. Holling and published by the International Institute for Applied Systems Analysis (Holling, 1978).

The Adaptive Environmental Assessment and Management (AEAM) approach involves the construction of a dynamic, computerized, simulation model which explicitly accounts for interactions between compartments of the system under study.

The inner workings of the simulation model are mathematical relationships, based on current understanding of lake-watershed ecosystems. The model output, in the form of projections of values for selected environmental indicators, provides information on the development capacity of the lake under consideration.

A key feature of the AEAM model is its ability to simulate the environmental consequences of different management strategies. Tangible measures of environmental impact make it possible to evaluate the options. With this type of analysis, a lake may have several acceptable levels of cottage development, depending on the specific management policy selected.

2.2 APPROACH TO DEVELOPING THE SIMULATION MODEL

The Integration Component scientists used the AEAM approach in constructing the Ontario Lakeshore Capacity Simulation Model. The method involves a co-ordinated, interdisciplinary approach that takes into account the needs and concerns of the ultimate users. Inherent in the process are short, intensive, modelling workshops which, for the Lakeshore Capacity Study, included scientists, planners, resource managers and policy advisors from the three participating ministries.

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* This chapter is based on the report Lakeshore Capacity Study, Integration (Teleki and Herskowitz, 1986).

Consultants from Environmental and Social Systems Analysts Ltd. (ESSA) were engaged to give clear direction to the modelling process. They acted as facilitators at the workshops and also designed and coded the initial Ontario Lakeshore Capacity Simulation Model (OLCSM).

Integration Component representatives also held a series of technical meetings with component managers and their senior staff, to resolve technical problems and to ensure consensus regarding the concepts and relationships expressed in the OLCSM equations.

2.3 ONTARIO LAKESHORE CAPACITY SIMULATION MODEL

The Ontario Lakeshore Capacity Simulation Model (OLCSM) is a type of simulation model sometimes referred to as a "state-transition discrete-time" model, because the state of the system is described by variables that are changed at specified intervals of time. All natural events, such as precipitation, lake discharge and growth of the fish population, are implemented from the start of the simulation. Random variation is applied to the annual nutrient loadings from precipitation and drainage, within the range observed in field studies.

Existing cottages are "built" at the start of the simulation. At any time thereafter, in response to a user-specified construction rate, cottages are built along the lakeshore on existing vacant lots and on lots in the proposed subdivision.

The inner workings of the model that provide the system dynamics (rules for change) describe in mathematical terms those processes and linkages between variables that are necessary for the prediction of environmental impact (Figure 2). These rules are primarily a product of functional relationships derived from the Lakeshore Capacity Study research, with supplementary data from the scientific literature and, where necessary, intuitive understanding of the lake-watershed system by experienced scientists. The levers represent the management strategies (actions) that can be tested with the model. The dials depict the measures of the state of the system (indicators). Values for the indicators are calculated at each time-step during the period of the simulation.

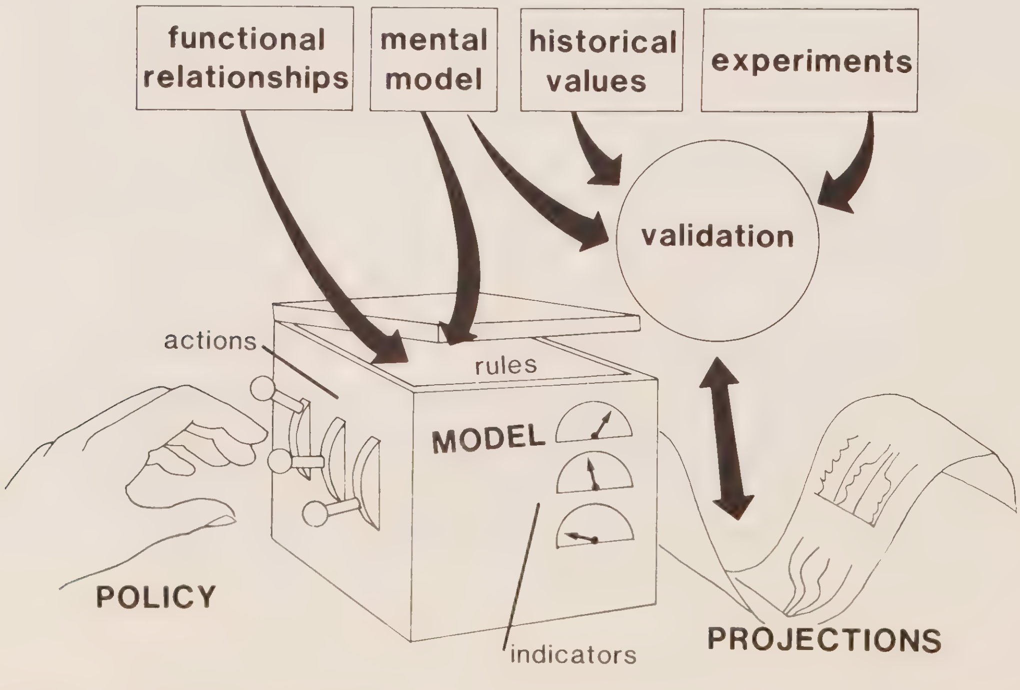

Figure 2. Modelling terminology

The OLCSM is designed to provide an objective measure of selected aspects of the environment, for a particular lake, under a given set of conditions. When new cottages are added to the existing stock, some change in the environmental indicator values is likely to occur. The amount of change that is considered acceptable is defined by the user, in the context of approved policies and standards.

2.3.1 SPATIAL AND TEMPORAL BOUNDARIES

A single lake, surrounded by a band of shoreland 50 metres deep, is the spatial unit selected for the OLCSM. Most of the submodels apply to an entire lake. However, smaller spatial units are needed for the Wildlife-Habitat submodel, so they are created by dividing the lakeshore and littoral zone into segments of uniform width. These wildlife segments are also used to identify the location of cottages and cottage lots.

Nutrient loadings from upstream and the immediate watershed are included in the model. Development on upstream lakes, land use within the watershed and size of the watershed are considered in the calculation of water quality indicator values. The model is structured so that chains of lakes can be evaluated. Beginning at the headwaters, the nutrient outflow from one lake is treated as nutrient input to the next lake downstream.

The time-step represents the increment of time between model calculations. In the present version, all submodels except Water Quality employ an annual time-step. In the Water Quality submodel, a seasonal time-step is preferable because phosphorus mass balance parameters can be estimated more accurately from seasonal data.

The length of each simulation is determined by the needs of the user and the implications of the data. For predicting the impact of cottage development, a 20-year simulation may be an appropriate period for planning purposes, but any time period can be selected. In order to simulate the retention of sewage-derived phosphorus in the soil around the septic system, it is essential to ensure that the time horizon for the simulation extends beyond the length of time the soil will retain its phosphorus-binding capacity.

2.3.2 INPUT AND OUTPUT (INDICATORS)

The submodel equations are designed to use data from existing records, such as the Ministry of Natural Resources creel census, maps and aerial photographs. The input consists of lake-specific data, such as the lake morphometry, lakeshore forest and littoral zone types around the lake, and locations of existing cottages, existing vacant lots and proposed cottage lots.

The model output consists of values for a set of indicator variables which function as gauges, or measures, of the response of the system. These values are plotted as graphs, or printed in tabulated form, for a user-specified interval of time and length of simulation. The indicators most useful to model users are those listed in Table 2.

Table 2. Principal indicators in the OLCSM

Number of lakeshore cottages
Total number of annual user-days for lake
Summer dissolved oxygen minimum
Winter dissolved oxygen minimum
Seasonal and mean ice-free lake phosphorus concentration
Algal biomass over ice-free months (wet weight)
Zooplankton biomass over ice-free months (dry weight)
Proportion of water samples with *Pseudomonas aeruginosa* pathogens
Maximum Sustained Yield (MSY) for smallmouth bass and lake trout
Fish carrying capacity of lake (by species)
Fish biomass (by species)
Fishing effort (by species and season)
Fish harvest (by species and season)
Population indices for small mammals and songbirds
Nesting habitat loss for hawks and loons
Mink activity index and disturbance factor

2.3.3 MANAGEMENT STRATEGIES (ACTIONS)

Selected management strategies, termed actions, may be introduced to mitigate the environmental impact of development. These actions pertain to parts of the ecosystem that can be managed and to modifications of the proposed development. They correspond to the controls a manager might exercise as part of a management policy. Actions can be taken to affect any aspect of the lake-watershed system. For example, changes may be made in the proposed plan of subdivision, the type of cottage access, the method of sewage disposal, or the length of the summer or winter fishing season. Management strategies included in the OLCSM are listed in Table 3.

The model is designed to indicate the future repercussions of various management options, to assist the user in choosing the most appropriate course of action for a particular lake. When several management scenarios have been simulated, the indicators from different scenarios can be compared and the most effective actions retained as a pool of viable management options.

2.3.4 SUBMODELS AND THEIR LINKAGES

The Ontario Lakeshore Capacity Simulation Model is divided into four subsystems, or submodels: Land Use, Water Quality, Fisheries and Wildlife-Habitat. Cottage use, identified as the major source of environmental impact from lakeshore cottage development, is predicted in the Land Use Submodel. Output from this submodel is then used in the Water Quality, Fisheries and Wildlife-Habitat submodels that calculate the environmental impact.

The Water Quality Submodel has two parts: one predicts the lake nutrient level, or trophic status, plus related water quality indicators, such as oxygen levels and transparency; the other is concerned with the health-related microbiology of lake water.

The Fisheries Submodel calculates the amount of fishing pressure, harvest and fish stock in a lake, in response to changes in the number of cottages, level of cottage use and number of anglers. The two sport-fish species included in the model are lake trout and smallmouth bass. The submodel is designed so that the lake trout are affected by reduced oxygen concentrations in the lake, whereas

Table 3. Management strategies in the OLCSM

LAND USE (COTTAGES)
Regulate cottage density
Require staged development for new subdivisions
Negotiate cottage construction rate
Discourage cottage conversion from seasonal to permanent use
Control type of access
Control size of common beach
Modify subdivision design to avoid interfering with critical wildlife habitat (e.g. adjust location of road, lot or beach)
WATER QUALITY
Limit phosphorus concentration in lake
Specify sewage system type and location for cottages
Require reduction of phosphates in sewage
FISHERIES
Shorten summer and/or winter angling season
Limit summer and/or winter catch
Encourage anglers to fish for underutilized species
Control public boat access
WILDLIFE-HABITAT
Reserve critical areas for deer, mink, loon, hawk and herptile habitat
Encourage cottagers to retain vegetation on lakeshore lots
Vary acceptable change in wildlife habitat

the smallmouth bass population changes in response to habitat alterations in the littoral zone.

In the Wildlife-Habitat Submodel, the impact of cottages on wildlife is calculated indirectly from changes in shoreland vegetation due to the development of cottage lots. This "habitat approach" emphasizes wildlife habitat disturbance. The wildlife included in the model are small mammals, songbirds, mink, loons, hawks, deer and turtles.

The linkages between the submodels represent the key interactions necessary for prediction of impact. The Land Use Submodel, for example, generates cottage-day and user-day data for all other submodels because these variables are a measure of cottage development, the source of the impact. Data on oxygen and lake phosphorus concentrations generated by the Water Quality Submodel and passed to the Fisheries Submodel reflect the linkages between water quality and fish biomass.

The overall framework for the OLCSM is illustrated in the flow diagram (Figure 3). Mathematical formulations and a fuller description of each submodel appear in the Integration report.

The methods of prediction used in each submodel of the OLCSM, which are drawn largely from the research results of the other Lakeshore Capacity Study components, are described in Part B (chapters 4-8) of this report.

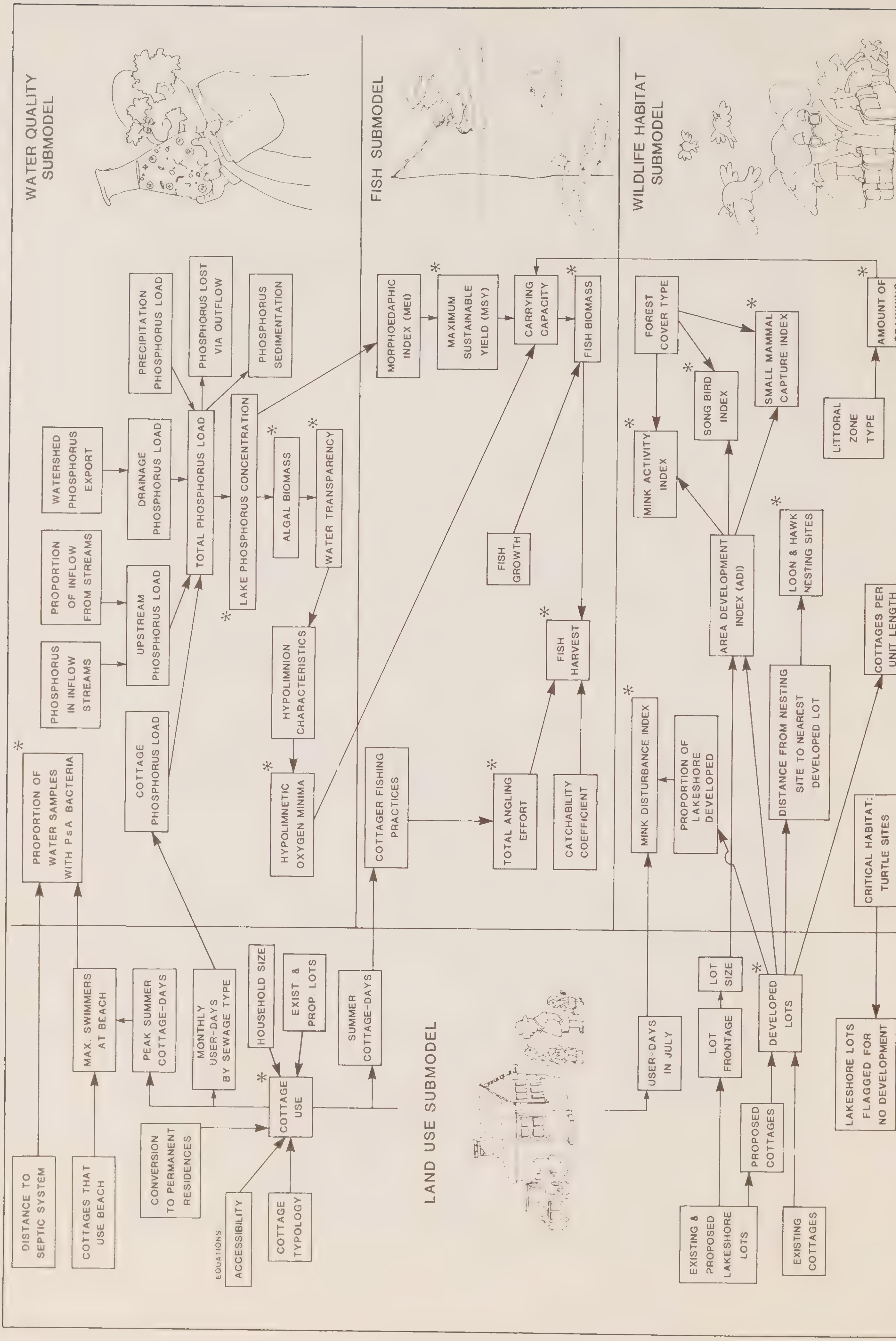

Figure 3. Flow chart of simulation model

3. POTENTIAL APPLICATION OF THE ONTARIO LAKESHORE CAPACITY SIMULATION MODEL

3.1 PRACTICAL APPLICATIONS IN THE PLANNING PROCESS

The aim of the Lakeshore Capacity Study was to produce a predictive model that could assist planners in the evaluation of subdivision proposals for cottage development on the shores of inland lakes. The Ontario Lakeshore Capacity Simulation Model not only fulfills this need but also has other potential applications in planning and resource management.

By providing predictions of the cottage capacities of different lakes, the model can assist regional planners in resolving demands for conflicting uses. While anglers may wish to have all lake trout fisheries protected, this may not always be feasible. Cottages may have to take precedence on some lake trout lakes. On others, it may be possible to resolve conflicts by introducing mitigative measures. How effective these measures may be can be tested by applying the simulation model.

The Land Use Submodel can be employed to predict where the greatest demand for cottages as principal residences is likely to occur. Such knowledge can assist planners in framing appropriate official plan policies and zoning regulations concerning cottage development. In addition, model predictions of the impact of different lot sizes on various wildlife species can assist in determining appropriate zoning standards for lot size.

Based on the location of the lake and the type of access, the Land Use Submodel can predict the user-days associated with cottages on a particular lake. Such a quantitative measure of the implications of different types of access on cottage use can assist planners in developing appropriate transportation policies in areas with inland lakes.

Whether roads to cottages are open in winter affects the lake trout fishery. For a specific lake, the simulation model can predict the effects of open or closed roads on the lake trout fishery, to assist planners and resource managers in recommending suitable snow clearing programs.

3.2 SOME CHARACTERISTICS OF THE ONTARIO LAKESHORE CAPACITY SIMULATION MODEL RELEVANT TO ITS PRACTICAL APPLICATION

While it is not the purpose of this report to explore the question of how the Ontario Lakeshore Capacity Simulation Model might be applied in the Ministry of Municipal Affairs, the description of the model would be incomplete if it failed to address in a general way some of the OLCSM characteristics that are relevant to its practical use.

The methods of prediction in the OLCSM were developed by researchers in the three ministries participating in the Lakeshore Capacity Study. Similarly, the

relevant data bases are lodged in these ministries. Therefore, input data for the OLCSM should be provided by the appropriate ministry. For example, the Ministry of Municipal Affairs should be responsible for the input data to predict cottage use; the Ministry of the Environment for data to predict the impact on water quality; and the Ministry of Natural Resources for data to predict the impact on fisheries and wildlife. Interpreting the predictions, keeping abreast of new scientific knowledge and monitoring the impact of new lakeshore development would also be the responsibility of the appropriate ministry.

The adaptive environmental assessment and management (AEAM) approach inherent in the Ontario Lakeshore Capacity Simulation Model works best when good liaison exists among planners, resource managers, scientists and policy advisors. While there are various ways of introducing the OLCSM, one approach might be to establish a small technical group, with representation from each of the three ministries. Together, the members of such an interdisciplinary team would have the knowledge necessary to operate the model, interpret the predictions and provide technical advice.

The OLCSM is not intended to be immutable. The present version of the model is capable of predicting the environmental impact of lakeshore cottage development and testing the effects of various mitigative measures. The next version should incorporate some or all of the refinements identified in the Integration report, to improve the accuracy of the predictions. The testing associated with refinements should include extreme situations, to ensure that the model is responsive to the full range of conditions in the Study Area. Subsequent versions of the OLCSM should reflect advances in scientific understanding of lake ecosystems and knowledge gained from monitoring the actual impact of new lakeshore development.

Finally, to view the OLCSM in proper perspective, it is essential to recognize that the methods of prediction are based on probabilities and that environmental predictions will always be based on incomplete scientific knowledge. While it may be tempting to seek greater precision, the crucial concern for planners is not whether the predictions are perfect but whether they are the best estimates of the future environmental repercussions of lakeshore cottage development. The OLCSM provides the best predictive methods presently available.

3.3 PREREQUISITES FOR OPERATING THE OLCSM

The operation of a computerized model, such as the Ontario Lakeshore Capacity Simulation Model, requires computer hardware, computer software and trained staff. Each of these essentials is described briefly in this section.

The OLCSM operates under a general purpose, time-sharing computer system. The model is controlled by the Simulation Control System (SIMCON) computer program, which handles input, output and record-keeping.

A computer terminal link to the computer system provides access to the model. With a graphics terminal, line graphs as well as tabulated output can be produced.

Operating instructions for the OLCSM are set out in the User's Manual (Teleki and Herskowitz, 1982). A "user-friendly" instruction program may be desirable for later versions of the OLCSM, to prompt the user regarding each entry.

To identify the staff skills required, a clear distinction must be drawn between operating the model, interpreting the predictions and refining the model. As the OLCSM responds to simple commands, a few weeks training is sufficient to learn what information to assemble, where to find it, how to enter it into the computer and how to generate predictions.

The predictions should be interpreted by specialists who have appropriate knowledge and experience. For example, the cottage use forecasts should be interpreted by planners in the Ministry of Municipal Affairs; water quality predictions by limnologists and microbiologists in the Ministry of the Environment; and fisheries and wildlife predictions by biologists in the Ministry of Natural Resources. One of the advantages of the OLCSM graphic output is that non-specialists can observe the trend lines formed from the predicted annual values of key indicators. However, only the specialists are qualified to interpret the predictions and to judge the effectiveness of mitigative measures.

Model refinement involves changes in the programming and coding. This demands both an understanding of the computer system and experience with simulation and environmental models.

PART B. RESEARCH RESULTS

4. LAND USE COMPONENT
5. TROPHIC STATUS COMPONENT
6. MICROBIOLOGY COMPONENT
7. FISHERIES COMPONENT
8. WILDLIFE COMPONENT

4. LAND USE COMPONENT

The goal of the Land Use Component planners and social scientists was to develop a method of predicting cottage use, for lakeshore cottages on a specific lake, prior to actual development.* The units of measurement chosen to describe the level of cottage use were termed cottage-days and user-days, where one "cottage-day" represents one cottage in use for one day, while one "user-day" represents one cottage in use for one day by one person.

Cottage use was important to the Trophic Status model because the number of people using the cottages affects the amount of phosphorous contributed to the lake and, in turn, water quality. Similarly, cottage use was important to the Fisheries Component net productivity model since the number of occupied cottages affects the number of anglers and, in turn, the amount of angling pressure exerted by cottagers on the fishery. Cottage use was also relevant to wildlife, as human activities associated with cottaging change the wildlife habitat on the lakeshore and, in turn, wildlife population and species.

4.1 EXISTING MODELS

A search of the literature in 1977 revealed no models that were specifically relevant to the task of predicting cottage use prior to development. While considerable work had been done relating to the use of cottages, much of it was descriptive rather than quantitative. Even in the latter category, while a number of studies produced estimates of annual cottage use, they did not identify the factors associated with different levels of use. Two exceptions are worth noting.

In the early 1970's, the Upper Great Lakes Regional Commission funded a study of lake property owners in northern Wisconsin (Klessig, 1973). The purpose was to learn more about the lake property owners, in order to determine their potential role in lake management. Reasons for variation in the time spent at the lake property were identified as the number of improvements on the property, age of the owner and location (rural or urban) of the owner's permanent residence. These three factors together explained 43% of the variance in the annual number of days at the lake.

In 1970, the College of Agriculture at Pennsylvania State University undertook a study of environmental quality aspects associated with seasonal home communities (Gamble et al., 1975). Most of the communities studied were located on lakes, with a few on the ocean or a bay. One of the objectives was to determine the attitudes of seasonal home occupants regarding environmental quality. Regression techniques were used to explain the reasons for the variation in annual use of the second home. The most significant variable was the seasons of the year the home was used — for each additional season, the use of the home increased by nine days. Also, older families and families who were willing to accept restrictions on their activities used their seasonal homes more. Lower use was associated with excessive noise, poor water quality and less emphasis on the quality of the home and community. Together, these variables explained 26% of the variation in seasonal home use.

* This chapter is based on the report Lakeshore Capacity Study, Land Use (Downing, 1986).

Some of the factors identified in these two studies as being associated with different levels of cottage use might assist in estimating the use of existing cottages. However, as explained in the following section, it was not feasible to employ them in attempting to predict cottage use at the planning stage, prior to actual development.

4.2 APPROACH TO DEVELOPING A MODEL TO PREDICT COTTAGE USE

The first step in developing a method of predicting cottage use prior to cottage construction was to recognize the stringent constraints on the choice of variables. As there would be no cottages or cottagers, no information would be available regarding improvements on the lot; number of seasons the cottage was occupied; family size, age or income; or attitudes and perceptions regarding noise or water quality.

The second step was to identify the kinds of information that would be available when a proposed subdivision was submitted for approval. The two major items were location of the subdivision and type of access to the cottage lots. Each of these was examined in the light of its potential as a predictor of cottage use.

The question of access offered several possibilities. If the time required to reach the cottage was a factor affecting cottage use, then any aspect of accessibility that would tend to increase or decrease the travel time might be relevant. One variable was the type of access, as some cottages had water access only. Another was the quality of the road. This could be measured in terms of road surface, as hard-surfaced roads had higher standards not only for surfacing but also for road width, curves and gradients, each of which could have some influence on travel time.

Given the location, it would be possible to measure the distance travelled on poor roads to reach the cottage. Similarly, the distance to urban centers in the region could be measured. Proximity to an urban center might tend to encourage longer stays at the cottage, due to the ease of replenishing supplies.

The 1978 Survey of Lakeshore Residents was designed to collect data from a representative sample of cottages on inland lakes in the Study Area, as a basis for model development. In total, 3,279 cottages received drop-off, mail-back questionnaires.

Information on cottage use pertained to the previous twelve months. The questionnaire included a calendar on which cottage owners were asked to record the number of people at the cottage on each day it was in use. From this detailed information, both cottage-days and user-days could be calculated, on a daily basis, for each cottage and each lake.

As the Lakeshore Capacity Study was concerned with the impact of cottages on the environment, it was essential to include cottages used all year as well as those used intermittently. Thus, the term "cottage" was used in a broad sense to include all single-family residential buildings located on the lakeshore or in close proximity to a lake. As most lakeshore development in the Study Area was single-tier development, most of the cottages had lake frontage.

Relevant information on the road network, type of cottage access and distance to urban centers was assembled in the summer of 1979 for the 1,139 cottages from which survey responses had been received. The data bank was completed in 1979, ready for the modelling process to begin in 1980.

4.3 RESEARCH RESULTS

4.3.1 COTTAGE USE

In general terms, the pattern of cottage use in Ontario was well known. It was widely recognized, for example, that summer was the period of greatest use, that winter use was increasing and that many cottages built originally for seasonal use were now being used all year. The 1978 Survey of Lakeshore Residents provided much more specific information for the Study Area. In particular, a record of the intensity of cottage use was obtained from the cottagers.

Of particular interest was the information on the number of days per year the cottage was used. It seemed logical to assume that the carrying capacity of the lakeshore would be greater for cottages used only part of the year than for those used all year. How much greater would depend in part on the amount of use.

A consistent seasonal pattern of cottage use was evident. Based on the monthly average for all cottages, the period of highest use extended for only two months, July and August; the period of lowest use extended for six months, from November to April. Transitional months occurred both in the spring (May and June) and in the fall (September and October).

The seasonal pattern of use was relevant to the analysis and prediction of cottage impact on the environment. It was apparent, for example, that cottagers could not be a major source of fishing pressure during the winter season, as few cottages were occupied at that time of year. Similarly, human disturbance of loons during their nesting period in June was probably limited, as cottages were not heavily used until July.

The daily use data (Figure 4) confirmed that there was a consistent weekly pattern of cottage use, not only during the summer but also throughout the rest of the year. More cottages were used on weekends than on weekdays. Further, more cottages were used on long weekends, such as the August Civic Holiday, than on other weekends.

While this information provided an illuminating overview of cottage occupancy, it did not link the use patterns to individual cottages. To do this, a statistical analysis was undertaken to develop a typology of cottages.

Figure 4. Percentage of cottages occupied daily during the year

4.3.2 COTTAGE TYPOLOGY

The initial purpose of the cottage typology was to classify the cottages according to their predominant use patterns, so the survey findings would provide a more realistic reflection of actual cottage use. It was known that the level of cottage use varied considerably, so it was not surprising that the overall average seemed unrealistically high to some cottage owners. Further, it was considered highly unlikely that every cottage was unique, so it was assumed that the cottages could be distinguished by their different use patterns and grouped accordingly. Cottage owners would then be able to identify a pattern of use similar to their own and the average annual use for that group of cottages would be much more likely to approximate their actual level of use.

The cottage typology analysis identified five mutually exclusive groups of cottages, each with similar use patterns. These are described below.

Group 1 ***Summer (July and August)***
Cottages used intensively during the summer (July and August) but much less at other times.

Group 2 ***Weekends***
Cottages used primarily on weekends in the spring, summer and fall.

Group 3 ***Long Weekends***
Cottages with low levels of use, except on long weekends in the spring, summer and fall.

Group 4 ***Extended Summer***
Cottages with high levels of use in the spring, summer and fall.

Group 5 ***All Year***
Cottages used all year, as a principal residence.

The five cottage typology groups had different average levels of annual cottage use and different average household sizes (Table 4). Cottages used all year, as a principal residence, were occupied about four times as much as those used predominantly in the summer (89 days) or on weekends in the spring, summer and fall (81 days). On average, cottages used all year had smaller households (2.56 persons) than those used predominantly in the summer (3.69 persons).

Table 4. Cottage use and household size for five cottage typology groups

Cottage Typology Group	Percent of Total Sample	COTTAGE USE Average Cottage-days per Year	HOUSEHOLD SIZE Average Persons per Cottage
Summer (July and August)	27.7	88.7	3.69
Weekends	28.0	80.5	3.60
Long Weekends	23.9	39.3	3.39
Extended Summer	7.8	175.2	2.64
All Year	12.6	363.1	2.56
All Cottages	100.0 (N = 1139)	116.03	3.07

4.4 MODELS TO PREDICT COTTAGE USE

The Land Use Component group considered several methods of predicting cottage use. One simple method, for example, was to assume that every cottage might be used all year. However, this "worst-case" assumption would drastically overestimate the actual level of cottage use, as the average use of a cottage in Muskoka-Haliburton (116 days during the year) was less than one-third of the maximum 365 days.

The implications of high predictions were considered. As the water in the lake is adversely affected by cottage use, the simulation model would respond to high cottage use estimates by forecasting lower water quality. In response to such a forecast, stringent limits would be placed on the creation of cottage lots, in order to preserve environmental quality. If this were to occur on many inland lakes, the effect on the total supply of cottage lots could be immense and the possibility of locating a cottage on the lakeshore could become more and more remote. In view of the implications, the "worst-case" approach was rejected.

Three methods of prediction were developed, which were termed the Averaging, Accessibility and Cottager models.

4.4.1 AVERAGING MODEL

In the Averaging Model, it is assumed that the average number of cottage-days recorded in the 1978 Survey of Lakeshore Residents would apply to all existing and proposed cottages. Thus, for a specific lake, total annual cottage use would be estimated by multiplying the number of cottages by 116 cottage-days.

The advantage of this method over the "worst-case" approach is its recognition of actual levels of cottage use. The predictions would tend to be realistic when averaged for all lakes.

The disadvantage of the Averaging Model is its inability to take into account variations in the average annual cottage use on different lakes and variations in the annual use of cottages on the same lake. Therefore, predictions of cottage use for a specific lake, using the Averaging Model, could be too high or too low.

4.4.2 ACCESSIBILITY MODEL

The predictive ability of the Accessibility Model is based on physical factors which measure the ease of access to the cottage. The major predictor is travel mode, which incorporates the type of access in summer and in winter. The other two predictors are the distance travelled on poor roads and the distance from the cottage to the nearest major urban centre in the region. Together, these variables explain 34% of the variance in annual cottage use.

Where the type of access is the same for each lot, the prediction of annual cottage-days for the subdivision is calculated by multiplying the predicted annual cottage-days per cottage by the number of lots (cottages) in the proposed subdivision. Where the type of access is different for some lots, those with the same type of access are grouped, a prediction is made for each group, and the separate predictions are aggregated, to provide a total for the whole development.

Conversion of the predicted cottage-days to user-days is accomplished by multiplying the predicted number of cottage-days for the subdivision by the average number of persons per cottage for the entire sample of cottages in the 1978 Survey of Lakeshore Residents. This "global" average (3.07 persons) is used for every subdivision.

The conceptual framework for the Accessibility Model is outlined in the flow diagram in Figure 5. The mathematical formulation of the model is provided in the Lakeshore Capacity Study Land Use report.

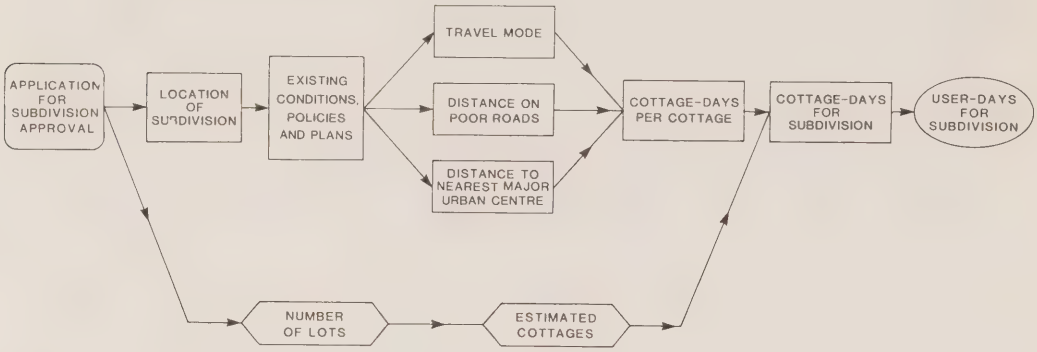

Figure 5. Accessibility Model: a conceptual diagram linking cottage accessibility and cottage use

The major advantage of the Accessibility Model is its ability to take into account variations in the average annual cottage use between lakes, by using as predictors accessibility characteristics associated with cottage use. Similarly, variations in cottage use between cottages are taken into account by predicting separately for groups of cottages with different types of access. Thus, the predictions for specific lakes are likely to be more accurate than those from the Averaging Model.

Another advantage of the Accessibility Model is that it stretches into the future as far as possible by employing Official Plans and Transportation Plans as a source of information for coding all the variables. In this way, proposed new roads, road improvements and realignments are built into the model to the degree to which they can be predicted at the time the model is applied.

The Accessibility Model is objective in the sense that all variables can be measured; relatively simple to apply because it employs only five predictor variables; and efficient in achieving good results with minimum effort.

4.4.3 COTTAGER MODEL

The pattern of cottage use recorded for each cottage in the 1978 Survey of Lakeshore Residents can be thought of as the end result of all the decisions made by cottagers regarding their use of the cottage. When viewed in this way, it is evident that the five distinct cottage groups identified in the cottage typology provide a valid basis for predicting cottage use, as they reflect not only decisions which may be influenced by the socio-economic characteristics of the cottage owners but also decisions in response to changes in external factors, such as fuel costs.

Initially, the Cottager Model was based on the five groups in the cottage typology. However, as the regression coefficients for the three low-use groups Summer (July and August), Weekends and Long Weekends were similar, they were combined into one Seasonal group in order to simplify the equation. This reduced the total number of cottage groups to three, namely Seasonal, Extended Summer and All Year.

These three groups of cottages differ substantially in their average annual cottage-days, the dependent variable in the Land Use Component models. The Seasonal group averages only 71 cottage-days per year, while the All Year group averages 363 cottage-days per year (Table 5). Between these extremes is the Extended Summer group, comprising cottages which are occupied for 175 cottage-days, on average. These differences give the model its predictive power.

Table 5. Cottage use and household size for three cottage typology groups

Cottage Typology Group	Percent of Total Sample	COTTAGE USE Average Cottage-days per Year	HOUSEHOLD SIZE Average Persons per Cottage
Seasonal*	79.5	71.0	3.57
Extended Summer	7.8	175.2	2.64
All Year	12.6	363.1	2.56
All Cottages	100.0 (N = 1139)	116.03	3.07

* Includes Summer (July and August), Weekends and Long Weekends typology groups.

The Cottager Model includes two cottage typology variables, Extended Summer and All Year, plus three accessibility variables. Although all the regression coefficients in the equation are highly significant, the variables representing cottage groups are the most reliable predictors.

As the values for the cottage typology groups will not be known when the model is applied, they must be derived from one of the benchmark lakes. These are, by definition, lakes where the cottage typology is known, so the only lakes which qualify are those included in the 1978 Survey of Lakeshore Residents. In addition, each benchmark lake must have a sufficient number of respondents to allow each typology group to be represented. Accordingly, the pool of candidate benchmark lakes comprises 36 lakes.

To select a specific benchmark lake, the goal is to find the best possible match in cottage typology between the target lake and its benchmark lake. Objective criteria are provided in the Land Use report to assist in choosing an appropriate benchmark. Recognizing that the strongest predictor in the Cottager Model is the proportion of cottages in the All Year typology group, the criteria suggest ways of using planning policies and zoning regulations as sources of information with respect to seasonal versus permanent use.

The conceptual framework for the Cottager Model is outlined in the network diagram in Figure 6. The mathematical formulation of the model is provided in the Lakeshore Capacity Study Land Use report.

The predictive power of the Cottager Model is extremely high, theoretically. When the cottage typology is known, the model explains 92% of the variance in cottage use among the cottages in the survey sample. However, when benchmarking is introduced, which is required in practice, a drastic reduction in

accuracy occurs. While there is no way of measuring it precisely, the Cottager Model appears to predict cottage use within a range of accuracy similar to the Accessibility Model.

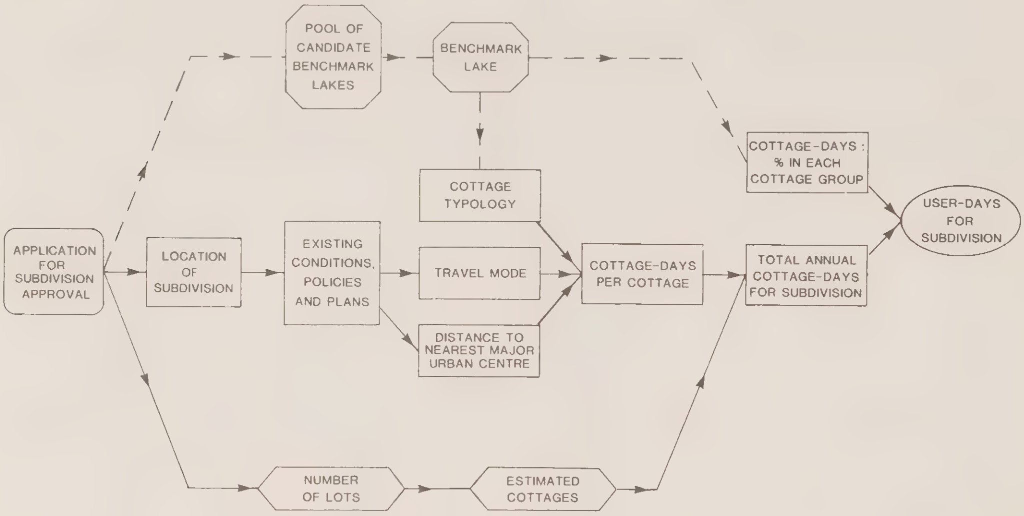

Figure 6. Cottager Model: a conceptual diagram linking cottage typology and cottage use

4.4.4 SELECTING THE BEST PREDICTION

The Land Use Component group tested predictions of cottage use generated by the three models, by comparing them with actual survey data. It was clear from the test results that the Accessibility and Cottager models performed better than the Averaging Model. As there was no statistical support for choosing between the Accessibility and Cottager models, the Land Use Component team recommended that both be employed. This would have the advantage of predicting on the basis of two different sets of information — one related to ease of access to the cottage and the other to patterns of cottage use.

The higher prediction of cottage use was recommended for use in the OLCSM. It more closely approximated actual cottage use where the travel mode was "car all year" and where the All Year typology group included a high proportion of the cottages. In other situations, the higher prediction tended to overstate cottage use, thereby providing some margin of safety while still remaining within a realistic range.

Particular care is required in designating the "travel mode" correctly, as it affects not only the predictive ability of the Accessibility Model but also the accuracy of the benchmarking which, in turn, affects the predictive ability of the Cottager Model. While several objective criteria are provided to assist in selecting an appropriate benchmark lake, some subjective judgement must be exercised. For this reason, application of the Land Use Component models requires professional knowledge of planning policies and regulations and their implications with respect to future development. Therefore, it is recommended that responsibility for the models remain with the Ministry of Municipal Affairs, which is responsible for administration of the Planning Act.

The derivation of the models is described more fully in the Land Use report. In addition, the Land Use Component models are described in the context of the Lakeshore Capacity Study as a whole in the Integration report.

5. TROPHIC STATUS COMPONENT

The Trophic Status Component research was designed to test and refine existing models for estimating the effects of lakeshore development on the trophic status of lakes.*

In essence, the trophic status of a lake is its capacity to sustain the growth and reproduction of plant and animal life. Phosphates, nitrates and other chemical substances, which leach into a lake from its watershed, serve as food or nutrients for a host of microscopic plants (phytoplankton). These plants drift in the lake water, providing food for minute animals (zooplankton and bottom fauna) which, in turn, are eaten by small fish.

Lakes with low concentrations of nutrients are said to be oligotrophic; those with high concentrations of nutrients are eutrophic; and those in an intermediate state are described as mesotrophic. Lakes located in the sparse soils and granitic rocks of the Precambrian Shield tend to be oligotrophic, while those in regions with deep glacial overburden and limestone bedrock are usually mesotrophic or eutrophic.

As phosphates and other plant nutrients accumulate in a lake, it may change from oligotrophic to mesotrophic and, subsequently, to eutrophic. Under natural conditions, such a process of change could take thousands of years. However, human activity on the lakeshore or in the watershed may accelerate the process if, for example, nutrients enter the lake in runoff from agricultural land or in seepage from waste disposal systems serving lakeshore cottage development.

5.1 EXISTING TROPHIC STATUS MODELS

The trophic status of a lake is usually determined by measuring specific water quality characteristics, such as chlorophyll *a* concentration (a surrogate measure for phytoplankton biomass), water clarity and the rate of oxygen loss from the profundal waters. It is well known that phosphorus, the algal nutrient present in shortest supply, controls the trophic status of most Precambrian Shield lakes because of its relationship with algal biomass and production.

The conceptual model (Figure 7) linking lakeshore development, lake morphometry, the hydrologic and phosphorus budgets, and the trophic status of a lake was originally outlined by Dillon and Rigler (1975). Inherent in the model was a critical untested assumption that lakeshore development using septic systems for waste disposal contributed phosphorus to the lake. In addition, many of the models were developed on the basis of data collected from a large number of lakes all over the world, which had a much wider range in trophic status than is typically found in lakes in the Precambrian regions of Ontario. As the existing models could not provide predictions with an acceptable degree of confidence over this smaller range in trophic conditions, it was necessary to undertake further model development, refinement and testing.

* This chapter is based on the report Lakeshore Capacity Study, Trophic Status (Dillon et al., 1986).

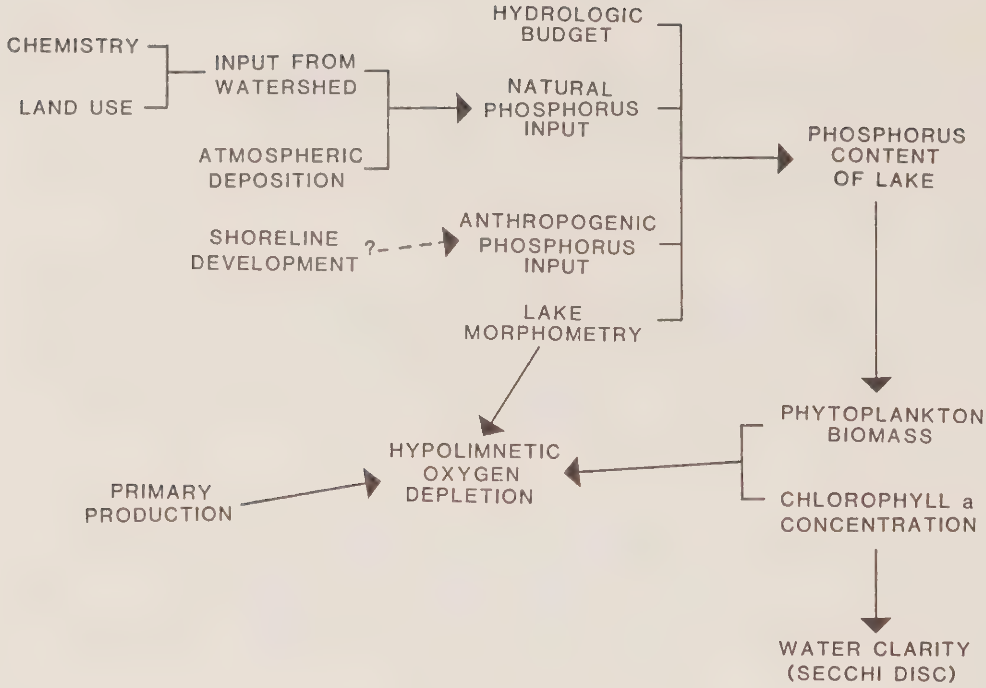

Figure 7. Conceptual model of the relationships linking phosphorus and the trophic status of lakes

5.2 STUDY DESIGN

The Trophic Status Component limnologists used two approaches in addressing the question of whether cottage development on the lakeshore contributed phosphorus to a lake.

In the first approach, the natural phosphorus and hydrologic budgets were measured for six lakes ("A" lakes) which exhibited a wide range in the potential significance of lakeshore cottages to the phosphorus budgets. Existing trophic status models were calibrated, using the two **undeveloped** "A" lakes; the models were then applied to the four **developed** "A" lakes. Differences between measured phosphorus concentrations in the developed lakes and phosphorus concentrations predicted from the calibrated existing models were assumed to be attributable to anthropogenic phosphorus inputs which, in turn, were quantitatively related to the extent of lakeshore development.

In the second approach, the intention was to compare the trophic status of nine lakes where lakeshore development was proposed ("B" lakes), before and after cottage construction. In fact, few of the anticipated changes occurred during the course of the Lakeshore Capacity Study, so such a comparison could not be made. Nevertheless, data collected from the "B" lakes contributed to the research, as the independent data set was used for testing models developed from the "A" lakes.

In addition, nine terrestrial watersheds ("export" watersheds) were selected for study, in order to broaden the range of geological characteristics considered.

5.3 APPROACH TO EVALUATING THE EFFECTS OF LAKESHORE COTTAGE DEVELOPMENT ON THE TROPHIC STATUS OF LAKES

As natural soil deposits around the lakes vary widely in their ability to assimilate phosphorus discharged from septic tile beds, the findings from a few study sites cannot be extrapolated with confidence to the entire lakeshore. Thus, direct measurement of the influence of lakeshore development on the trophic status of lakes is extremely difficult. To overcome this problem, the Trophic Status Component limnologists used two indirect approaches, involving comparisons between the developed and undeveloped lakes with respect to (i) their nutrient regimes and (ii) several biological characteristics.

Of the six intensively studied "A" lakes, two had little or no development on the lakeshore or in the watershed, while the other four had a significant amount of development. However, due to natural variations in the nutrient content of undeveloped lakes, a direct comparison of the phosphorus (or other nutrient) content of the developed and undeveloped lakes was not a valid approach for measuring the effects of development. Fortunately, conceptual links between lakeshore development and the phosphorus budget of a lake were well established. Hence, it was possible to formulate the conceptual model in mathematical terms, calibrate it using the undeveloped lakes, and then apply it to the developed lakes to ascertain the effects of lakeshore development on the phosphorus budget and thus on trophic status.

Several biological components of the developed lakes were compared directly with the same components of the undeveloped lakes. These biological components included phytoplankton, zooplankton and macrophytes. While natural differences in trophic status confounded the interpretation of community data, restricting the usefulness of this approach, direct comparison did allow implicit evaluation of the influence of lakeshore development on biological communities that were not related to changes in nutrient supply.

5.4 THE PHOSPHORUS BUDGET APPROACH

Factors that determine the total phosphorus concentration in a lake include lake morphometry, the hydrologic budget, phosphorus input and the phosphorus loss rate. The conceptual model linking these factors was presented in Figure 7.

The morphometry of each of the "A" lakes, including lake area, volume, mean and maximum depth, and length of the shoreline, was measured at the beginning of the study. The hydrologic budgets of the "A" lakes were measured directly for the four-year period June 1976 to May 1980. The components of the hydrologic cycle are related by the water balance equation, which states that the change in lake volume is equal to precipitation to the lake surface plus surface runoff into the lake minus outflow and net evaporation from the lake. Water balances were constructed for four consecutive 12-month periods, beginning in June 1976, and for the entire four-year period. Because each term in the water balance equation was measured directly, it was possible to evaluate the accuracy of the hydrologic budgets.

Total phosphorus inputs from all significant sources, other than those associated with lakeshore development, were measured for the same four-year period. The measured sources included atmospheric deposition and watershed runoff (Figure 8).

Loss of phosphorus by outflow from each lake was also measured, for the same four years. Loss of phosphorus by sedimentation within each lake required quantification but the rate of accumulation of phosphorus in the sediments of lakes is difficult to measure independently. It is possible to calculate the phosphorus sedimentation rate by difference between the total input rate and the loss rate attributed to outflow, if corrections are made for changes in phosphorus concentration in the lake. This method is suitable for the **undeveloped** lakes only, since the actual total input is unknown for the developed lakes. A general sedimentation rate model or an independent measurement of sedimentation rate was therefore needed.

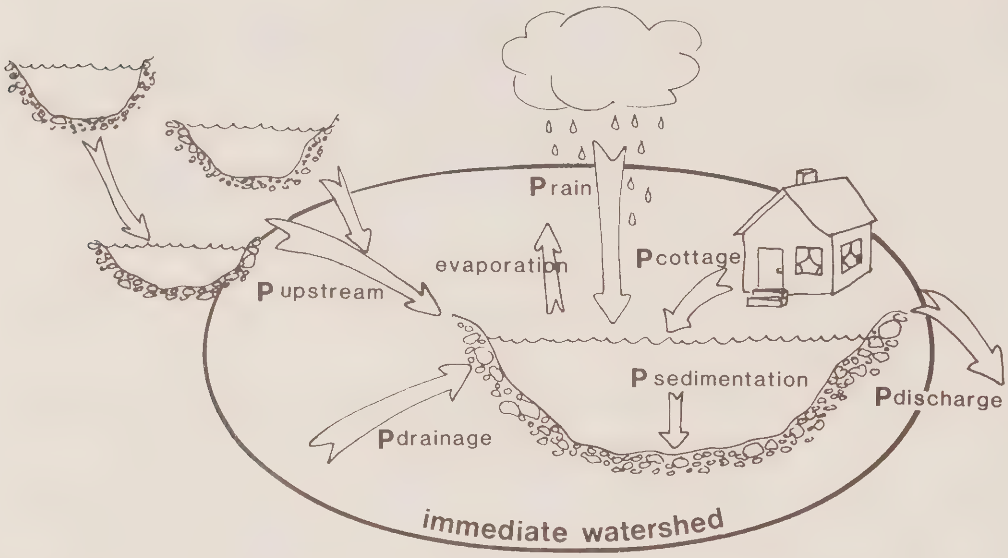

Figure 8. Phosphorus input and output

5.5 RESEARCH RESULTS

The major findings of the Trophic Status Component group regarding the hydrologic and phosphorus budgets of the "A" lakes are summarized below.

Hydrologic budgets, which were calculated for four annual periods (June to May) and for the entire four-year period (1976-80), balanced to within ± 10% for five of the six "A" lakes. This gave confidence in the individual supply and loss terms and in the use of the data for constructing chemical mass balances and calculating water replenishment times for the lakes.

A wide range in annual atmospheric deposition of phosphorus was apparent, indicating that long-term monitoring was essential in order to obtain reliable estimates. A seasonal pattern in the deposition of phosphorus was observed, with peak values occurring in the summer months. Deposition rates at different locations did not vary significantly.

The four-year phosphorus export (yield) for 30 watersheds varied widely from one watershed to another. The grand mean phosphorus export for metamorphic watersheds was somewhat higher than the mean based on previous studies. Year-to-year variability in yield was not significant compared to site-to-site variability.

Based on a literature review, the average phosphorus supply to sewage disposal systems was calculated to be 0.8 kilograms per person per year. Phosphorus retention in the sewage disposal system was estimated for different types of systems in use around the study lakes. Based on these estimates, phosphorus retention in septic tank and tile field systems was assumed to be zero unless information was available on the condition of the system and the type of soil, in which case a correction factor for phosphorus retention in the soil could be applied. Information on the levels of cottage use and the types of sewage disposal systems, obtained from the Land Use Component, allowed the potential anthropogenic phosphorus input to each study lake to be estimated. These figures ranged from zero to 83 kilograms of phosphorus per year.

Loss of phosphorus to the lake sediments was calculated for the undeveloped lakes in three ways: from measured phosphorus mass balances; from sediment trap data; and from information on the concentration of phosphorus in sediments and the sediment accumulation rate. When the results were compared, it was evident that sediment traps overestimated the phosphorus sedimentation rate, whereas the sediment chronology method provided results comparable to those from the mass balances method.

The Trophic Status Component scientists also investigated minor components of the phosphorus budget. Based on data from one lake, net phosphorus loss by fish harvest was estimated at 2% of the total phosphorus supply; phosphorus loss by emerging insects was negligible; and phosphorus supply from forest litterfall was estimated at 3%of the total phosphorus input.

Using phosphorus mass balance information, the relative importance of the various sources of phosphorus was evaluated. Inflowing streams varied widely in importance, providing from 0.4% to 69% of the natural phosphorus input to the lakes. Precipitation was the single most important source of phosphorus in four of the six "A" lakes, contributing 20% to 74% of the total natural supply (1976-80 mean values). Contributions from the ungauged watersheds were relatively small, averaging 15% of the total natural supply of phosphorus to the lakes. The loss of phosphorus to the sediments was always greater than the loss to the lake outlet, even when potential anthropogenic inputs were not considered. Finally, the potential contributions from anthropogenic sources of phosphorus ranged from zero to 55% of the total phosphorus supply. Substantial year-to-year variability in individual components of the phosphorus budgets indicated that mass balances must be measured for several years in order to develop and validate trophic status models.

The impacts of lakeshore development on communities of phytoplankton, zooplankton and macrophytes were evaluated on the "A" and "B" lakes. This was done by examining the correlations between parameters describing the state of the biological communities and indices of lakeshore development or by comparing the values of the parameters for sets of lakes with extremely different intensities of cottage use. When phytoplankton community parameters for the three most developed lakes and the four undeveloped lakes were compared, there was no significant difference between the two sets of lakes. A similar comparison for zooplankton community parameters (1977-79) showed no significant difference between lake sets. Further, no detectable difference was found in the composition, standing stock or nutritional status of macrophytes in eight lakes covering the full range of human use. Thus, there was no evidence of any direct relationship between lakeshore development and these aquatic biota.

5.6 MODELS TO PREDICT TROPHIC STATUS

The Trophic Status Component scientists started from a strong base in Phase III of the Lakeshore Capacity Study. A conceptual model and several mathematical models were already available to assist in predicting the effects of lakeshore cottage development on the trophic status of a lake. Accordingly, in Phase III, existing models were refined and new models were developed to strengthen and link individual parts of the conceptual model.

The resulting models include those for the prediction of: (1) important components of the hydrology of lakes and watersheds; (2) yield or export of phosphorus from terrestrial watersheds; (3) retention of phosphorus in the lake sediments; (4) concentration of phosphorus in a lake; (5) phytoplankton biomass and chlorophyll *a* levels; (6) water clarity (as measured by Secchi disc); (7) zooplankton biomass; and (8) depletion rate of oxygen from the profundal waters. These models are described in general terms below.

Existing models predicting the four principal hydrologic components (precipitation, runoff, evapotranspiration and evaporation) from readily obtainable hydrometeorological data were refined and tested on the "A" lakes and "export" watersheds. Precipitation at a given site could be predicted, using data from nearby meteorological stations. Of the five techniques investigated to predict runoff, the preferred method was the modified Tennessee Valley Authority (TVA) model because it allowed prediction of streamflow on a fine time-scale (daily) for any given set of climatic conditions. After calibration on the "A" lake watersheds, the modified TVA model was tested on the "export" watersheds and gave good predictions of daily flow in all years except 1977-78. Both basin evapotranspiration and lake evaporation could be predicted, using climatological models.

Two models were developed to predict the annual yield of phosphorus from terrestrial watersheds. In one model, 51% of the variance in the phosphorus yield of 30 watersheds was related to geologic, land use and hydrologic characteristics of the watershed. In the other model, 46% of the variance in the phosphorus yield was explained by geologic and land use characteristics alone.

The relationship given in Dillon et al. (1977) successfully predicted the annual retention of phosphorus from the areal water load, when tested against estimates derived from the phosphorus mass balances in two undeveloped lakes. The model was recalibrated for anoxic (oxygen deficient) lakes, resulting in an adjustment of the phosphorus settling velocity.

An existing model for predicting the concentration of phosphorus in lakes at spring turnover was refined to allow prediction of phosphorus in any season and to incorporate the relation between the concentration of phosphorus in the lake and that in the lake outlet. In lakes with no anthropogenic input of phosphorus, the predicted and observed mean (1976-80) concentrations of phosphorus during the ice-free season agreed, giving confidence in the use of the model in lakes with anthropogenic inputs of phosphorus. Incorporating thermal stratification in the model did not improve its predictive capability.

Using the revised phosphorus retention and concentration models, predicted and observed concentrations of phosphorus were compared for the four "A" lakes with lakeshore development and the fraction of the potential anthropogenic phosphorus input to the lakes was calculated. For three lakes, the predicted and observed phosphorus levels agreed, if it was assumed that 100%

of the anthropogenic phosphorus supply entered the lakes. However, for one lake, only 20% of the anthropogenic phosphorus supply must enter the lake for the observed and predicted phosphorus levels to agree. This may be due to differences in the phosphorus retention capability of the soils surrounding the lake.

Models were developed from the "A" and "B" lake data to predict phytoplankton biomass, corrected phytoplankton biomass (excluding diatoms) and the concentration of chlorophyll *a* (a correlate of phytoplankton biomass) during the ice-free season, from epilimnetic concentrations of algal nutrients (1976-80 mean values). Diatoms were excluded from the "corrected" phytoplankton biomass in part because many diatom species contain a relatively large empty space but also because of the difficulty in distinguishing between living and dead diatoms. Chlorophyll *a* and corrected phytoplankton biomass could both be predicted from the concentration of phosphorus and the ratio of the concentration of nitrogen and phosphorus. The phytoplankton biomass was related to the concentration of total organic nitrogen. Including total zooplankton biomass or herbivorous zooplankton biomass in the relations did not improve their predictive ability.

Water clarity (as measured by Secchi depth) of the "A" and "B" lakes during the ice-free season was controlled by the dissolved organic carbon content (or colour) of the lakes. Chlorophyll *a* concentration and phytoplankton biomass were not strongly correlated with water clarity.

Models for predicting areal hypolimnetic oxygen rates were tested. The Cornett-Rigler model (Cornett and Rigler, 1979), when recalibrated with data collected in this study, provided the closest agreement between predicted and observed depletion rates.

Total crustacean zooplankton biomass and herbivorous zooplankton biomass of the "A" and "B" lakes (1977-79 mean values) could be predicted from the whole-lake concentration of phosphorus during the summer stratified period. These predictions may be useful in future efforts to model fish biomass.

Acidic deposition, a result of the oxidation and hydrolysis of sulphur and nitrogen oxides released as a product of fossil fuel combustion or the smelting of sulphide ores, may be the single most significant environmental problem in eastern North America. Lakes on the Precambrian Shield are particularly sensitive to the deposition of these strong acids. As loading rates of acids in the Study Area are as high as those in any area of the world, the Trophic Status Component analysis was designed to ascertain whether nutrient - trophic status relationships were influenced by acidic deposition. The chlorophyll *a*, phytoplankton biomass and water clarity models were derived with consideration of both nutrient enrichment and acidification parameters. The analysis produced no evidence that the pH of the lakewater, which is an indicator of acidity, affected the nutrient - trophic status relationships in the study lakes.

Relationships among the models developed by the Land Use, Trophic Status and Fisheries components are described in the Integration report, in the context of the Lakeshore Capacity Study as a whole. Mathematical formulations of the trophic status models appear in the Trophic Status report, where their derivation is described more fully.

6. MICROBIOLOGY COMPONENT

The goal of the Microbiology Component research was to define and quantify the influence of lakeshore cottage development on the bacteriological quality of the lake water and infections among human users of the water.*

The need for adequate treatment of drinking water and sewage in a cottage development is well documented in the literature. In Ontario, the policies and practices of the Ministry of Health and the Ministry of the Environment minimize the possible presence of disease-causing organisms in the drinking water supply and the contamination of the environment with sewage. The Ministry of the Environment and the various medical authorities have an effective procedure for determining whether improperly treated sewage is creating a health hazard, thus requiring immediate remedial action. Therefore, the Microbiology Component group directed their Lakeshore Capacity Study research toward swimming-related illnesses that were less well understood and documented.

While human illness due to swimming has been relatively uncommon, several serious outbreaks of bacterial and viral infections have been reported. The types of human disease associated with swimming include gastrointestinal illness, wound infections, and infections of the eyes, ears, nose, skin and throat by a variety of bacteria and viruses. Of these, the ear infection *otitis externa* or "swimmer's ear" accounts for the majority of swimming-related illnesses. It is a painful ear infection requiring medical diagnosis and treatment.

Infectious microorganisms found in recreational waters can come from one of three sources: pollution by human or animal sewage, the washing of the skin of bathers, or the resident population of environmental microbes. The last is generally not controllable but seems to play a minor role in causing human disease. For the other two, the contribution of the infectious agent to the recreational waters depends on such factors as the level of illness in the contributing population, the number of carriers of the agent and the effective countermeasures introduced.

The probability of an ill individual or a carrier being present in small populations, such as the bathers from a few cottages, is correspondingly small. Under these circumstances, the occurrence of the infectious agent will be sporadic, so the normal indicator systems may fail. However, by determining the concentration of the pathogen directly, this problem can be alleviated. Accordingly, the research undertaken by the Microbiology Component scientists was designed to examine the relationship of the pathogen *Pseudomonas aeruginosa* to the ear infection *otitis externa*.

* This chapter is based on the report Lakeshore Capacity Study, Microbiology (Burger, 1983).

6.1 PSEUDOMONAS AERUGINOSA - OTITIS EXTERNA

Pseudomonas aeruginosa is the primary organism associated with the external ear infection *otitis externa*. According to the literature, *P. aeruginosa* occurs mainly in humans.

As an infectious agent, *P. aeruginosa* has been described as an opportunistic pathogen. It may be present without apparent illness or disease but if the resistance of the host is reduced by trauma, other disease, medication or predisposition, a secondary infection by *P. aeruginosa* can occur.

Although a direct correlation has not been established between ear infections and waterborne *Pseudomonas aeruginosa*, the reported incidence of *otitis externa* is higher among swimmers than non-swimmers. Ear infections due to *P. aeruginosa* may even occur with low densities of pathogens in the water.

When a swimmer submerges his head in water containing *P. aeruginosa*, a small but definable portion of water enters the ear canal, carrying the pathogen with it. This water, with its bacterial content, may either drain out or be dried out without any further action or it may moisten the ear canal contents and remain for a considerable time. The bacteria may also either settle out or adhere to the inner surfaces even though the water is removed. Once the *P. aeruginosa* has entered the ear, an infection can occur. However, the process by which the infection occurs is not clearly understood.

6.2 FIELD SURVEYS

The Microbiology Component field work included two types of survey. One involved regular sampling surveys of lakes to determine the distribution and densities of bacteria and the levels of important nutrients. These were designed to provide information concerning the effects of development on bacteriological water quality and on possible relationships between lakeshore development, bacterial densities and nutrient status. The other type of survey was an epidemiological investigation of *P. aeruginosa* - related ear infection among users of shoreline waters and beaches.

The beach studies involved several kinds of data. Information was gathered relating to the history of ear infection, swimming habits and water-based recreational activities of cottagers and swimmers. Samples of bathing beach waters and shoreline sediments were selected and analyzed for indicator bacteria and *Pseudomonas aeruginosa*. The ears of swimmers were swabbed prior to and just after swimming. Swab cultures were then analyzed for *P. aeruginosa* and the results from the pre- and post-swimming swabs compared.

Data from the lake and beach surveys were analyzed to identify relationships between the incidence of *otitis externa*, bacterial and chemical water quality, levels of human use of the water (swimmer load), and environmental conditions.

6.3 RESEARCH RESULTS

All the evidence from the Microbiology Component research suggested that a relationship existed between *P. aeruginosa*, the number of people using the beach, and the water temperature, but the relationship could not be validated statistically.

Based on the results reported in the literature and the findings of their study, Microbiology Component scientists placed values on the levels of occurrence of

Pseudomonas aeruginosa. About 10% of the normal healthy human population carry *P. aeruginosa* in their gastrointestinal systems. As a result, the pathogen is present in human sewage.

In north temperate surface waters which are not affected by humans, the *P. aeruginosa* levels are extremely low. Where human activity occurs, such as at a beach with public access, the pathogen can be measured. The levels of *P. aeruginosa* tended to increase as the number of beach users increased.

The time trend of *Pseudomonas aeruginosa* found in swimmers' ears followed closely that of *P. aeruginosa* found in the water. During the period mid-July to mid-August, when lake water temperatures rose to 24° or 25°C, *P. aeruginosa* was recovered more frequently.

While *Pseudomonas aeruginosa* is found in a low percentage of normal, healthy ears, Microbiology Component data showed higher percentages when measured after swimming. Up to 3% of the swabs from healthy swimmers' ears before swimming and up to 8% of the swabs after swimming were positive for *P. aeruginosa*. The only source of the new *P. aeruginosa* in the ears was water contaminated with the pathogen, which entered the ear while swimming.

The *otitis externa* ear infection occurred in 7% to 12% of the swimmers. The swimmers most likely to contract an infection were those in the age group 5 to 24 years, those who swam frequently, and those with a previous history of ear infection.

These data show that if the water in a swimming area contains *Pseudomonas aeruginosa*, swimming activity will introduce the pathogen into the swimmer's ear. Whether the *P. aeruginosa* will cause an ear infection will depend on a number of medical circumstances, which are beyond the scope of this study.

6.4 RECOMMENDATIONS

The importance of adequate treatment of the water used for drinking, washing and food preparation, and of the sewage from cottages, cannot be stressed strongly enough. In the examination of any proposal for development, these areas must be considered if the health of the people is to be safeguarded.

In this study, the research was restricted to one organism, *Pseudomonas aeruginosa*, and one disease, an ear infection called *otitis externa*. The model developed by the Microbiology Component (Figure 9) is supported by the general microbiological literature, by the general inclination of microbial ecology and by the findings of the Microbiology Component scientists. It is a qualitative model with some sections having quantitative data.

The results and the model provided a basis for the following recommendations of the Microbiology Component with respect to a *P. aeruginosa* water quality criterion for use in Ontario and criteria for the location and design of swimming areas in order to minimize the risks of *otitis externa* ear infections.

The incidence of ear infections among swimmers at a particular location increased noticeably when the 75 percentile concentration of *P. aeruginosa* exceeded 10 bacteria per 100 mL. This level of bacterial contamination can, therefore, be set as a water quality criterion. When the *P. aeruginosa* concentrations in a swimming area regularly exceed this level, the risk of contracting an ear infection is greater and the use of the swimming area should be restricted until the situation has been rectified.

A swimming area associated with a cottage development should be located upstream and away from possible sources of sewage and sewage impacted soil runoff. It should also be located away from shallow, organically rich, marshy areas which may be warmed by the sun or by a thermal input. In addition, it should be designed to ensure good flushing or dilution with water from outside the impacted area.

Swimmers should be dispersed over as wide an area as possible, to reduce swimmer loading, the *P. aeruginosa* levels in the water and, in turn, the likelihood of *otitis externa* ear infections.

The Microbiology Component work is described more fully in the Microbiology report. In addition, the research results are discussed in the context of the Lakeshore Capacity Study as a whole in the Integration report, in the section on the Water Quality Submodel.

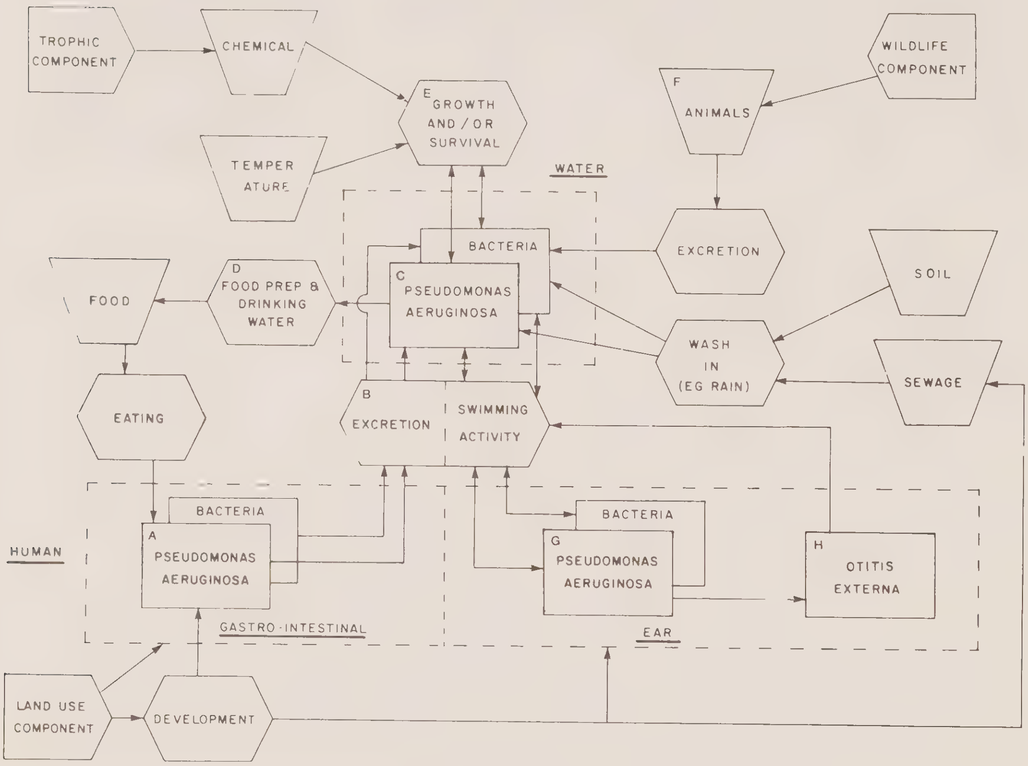

Figure 9. Flow diagram for *Pseudomonas aeruginosa* and its relationship to swimming activity and the ear infection *otitis externa*

7. FISHERIES COMPONENT

The construction and subsequent use of cottages and associated facilities on the lakeshore can affect the fisheries in several ways, including changing the trophic status of the lake; altering the spawning, nursery and feeding grounds of the fish; and increasing the fishing pressure. The goal of the Fisheries Component net productivity researchers was to develop a method of predicting the impact of lakeshore cottages on fisheries, as a result of fishing pressure.*

The production of sport fish and each intermediate link in the food chain depend ultimately on the supply of nutrients (Figure 10). Because phytoplankton require sufficient sunlight to support photosynthesis, the shape of the lake basin also affects trophic status. The fraction of the lake volume receiving sunlight is usually smaller for a deep lake than for a shallow one.

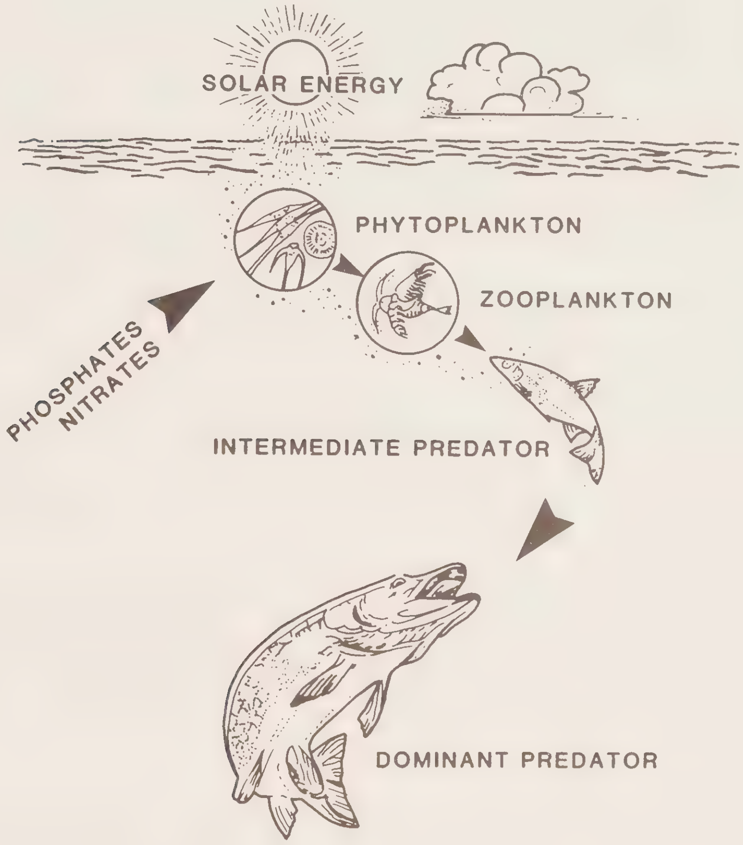

Figure 10. A typical aquatic food chain illustrating the connection between dissolved solids such as phosphates and nitrates, solar energy and, ultimately, the production of game fish (e.g.northern pike)

* This chapter is based on the report Lakeshore Capacity Study, Fisheries, Net Productivity: Index of Cottage Impact on Fisheries (McCombie, 1983).

7.1 EXISTING MODELS

A relationship between fish catch, water chemistry and geology was described by Ryder (1965). Based on a study of 23 lakes in Ontario, he found that the fish catch, or yield, was related to the ratio of total dissolved solids (an edaphic characteristic) and mean depth of the lake (a morphometric characteristic), which he called the morphoedaphic index (MEI). Total dissolved solids are a measure of the overall contribution of nutrient elements to the lakes from the watershed, while mean depth is an indicator of the fraction of the lake volume suitable for photosynthesis.

In the 1970's, Jenkins and Morais (1971) and Henderson and Welcomme (1974) found that fish catches were also strongly correlated with fishing pressure. Henderson and Welcomme combined the fish yield, morphoedaphic index and commercial fishing effort in a single model.

7.2 APPROACH TO EVALUATING THE IMPACT OF LAKESHORE COTTAGE DEVELOPMENT ON FISHERIES

Creel censuses were conducted on 46 lakes in the Muskoka-Haliburton area, during the summers of 1975-79 and the winters of 1976-79, to obtain information regarding anglers and their catch. The lakes were selected to give a representative sample of lake surface area, mean depth, total dissolved solids content, fish species and degree of cottage development on the lakeshore.

The creel census involved interviews with anglers on the lake. On each census day, the census-taker made one trip around the lake and interviewed all anglers on the route. Information was collected on the number of anglers in the party, number of hours fished prior to the interview, estimated total number of hours the party expected to fish that day, type of anglers (cottagers or non-cottagers), origin of anglers (home location), and the number of fish caught, released and sampled. When possible, fish length and weight were also recorded. From the summer of 1977 on, scale samples were collected to determine the age of the fish.

Anglers from cottages on the lake were separated from non-cottage anglers for analytical purposes, in order to estimate the fishing pressure exerted by each group, on each lake. For non-cottagers, the information on home location was analyzed to determine whether they came from distant points or communities near the lake.

As angling seasons for various species may differ with respect to opening and closing dates, which may also vary from year to year, an arbitrary number of days was assigned to each season. The summer fishing season was assumed to be May 1 to September 30 (153 days) and the winter fishing season January 1 to March 31 (91 days).

7.3 RESEARCH RESULTS

Analysis of the creel census data from the net productivity study lakes indicated that most fishing parties spent from two to four hours per day angling in summer, whereas in winter a considerable proportion spent up to six hours per day. The Fisheries Component biologists attributed the shorter fishing day in summer to the greater choice of aquatic recreational activities available in summer and to fishing parties including children, who have a short attention

span for angling. They suggested that the longer fishing day in winter might reflect difficulty in reaching the fishing site, relative comfort of the fishing hut and the amount of free time available to local residents in winter.

A consistent relationship was evident between fish catch during the summer and the morphoedaphic index. In addition, the summer catch of all fish species was related to summer angling effort.

The next step was to link angling effort to cottages. The number of anglers was calculated from the number of cottages on the lakeshore, by assuming an average number of anglers per cottage. Angling effort was then estimated by multiplying the number of anglers by the average number of hours per fishing trip, based on creel census data.

An indirect approach was used to estimate angling effort for non-cottagers. Assuming that a lake which was popular with cottagers was also likely to be popular with non-cottagers, the non-cottager angling effort was related to cottager angling effort. The correlation between cottager and non-cottager effort was found to be highly significant for the net productivity study lakes.

Estimates of winter angling effort were also derived indirectly, because many winter anglers could not be identified with points of origin on the lakeshore. Some came from nearby villages while others were from cities a considerable distance away. It was assumed that fisheries which attracted anglers in summer would also attract anglers in winter. The regression analysis showed a strong correlation between summer and winter angling on the net productivity lakes.

Winter angling differed from summer angling in terms of the people involved. Creel census data showed that, on average, cottagers accounted for 52% of the total summer angling effort but only 14% of total winter effort. Day-trippers and local residents formed the largest groups of anglers in winter on the net productivity lakes.

Some authors have reported that winter angling may be more effective than summer angling and may take a larger proportion of small, immature lake trout. On the net productivity lakes, this was not evident from creel census data, so an angler-hour of winter effort was considered to be equivalent to an angler-hour of summer effort.

7.4 NET PRODUCTIVITY MODEL

The morphoedaphic index combines water chemistry with lake morphometry and relates the trophic status of the lake, as measured by production of game and commercial fish, to the geology of the lake basin and watershed. The net productivity index incorporates this concept and adds angling effort as another predictor. The latter provides a link to cottage development, as angling effort is estimated from the number of anglers which, in turn, is derived from the number of cottages.

The net productivity index is designed to predict fish yield from the lake. This does not, in itself, indicate the number of cottages that can be built on the lakeshore without generating undue pressure on the fishery. However, when the predicted fish catch is compared with the maximum sustained yield, the effects on the fishery become evident.

The maximum sustained yield for the fish community is the maximum weight of two or more fish species which could be removed from the lake without altering

the kinds and sizes of fish caught. Similarly, the maximum sustained yield for a particular species, say lake trout, is the maximum weight of lake trout which could be removed from the lake year after year without significantly altering the ages and sizes of lake trout caught.

In the net productivity model, the weight of fish removed by anglers is predicted by a series of regression equations, derived from data obtained from anglers on 46 lakes in the Muskoka-Haliburton area plus data for ten lakes selected from the literature. The equations can predict:

(1) summer angling effort by cottagers,
(2) summer effort by anglers other than cottagers,
(3) total winter effort by all anglers, and
(4) total fish catch (yield) by all anglers.

In general, if the angling yield is greater than the maximum sustained yield, the fishery is considered to be overexploited. In contrast, if the fish catch is less than the MSY, the fish population can tolerate more fishing pressure.

While the community maximum sustained yield provides a good first approximation of potential fish production for a lake, exploitation of the fishery at this level is considered unwise. For one reason, fishing pressure may have a greater impact on some fish species than on others. For another, some year classes may be weaker than others. Accordingly, to provide a margin of safety, the Fisheries Component biologists recommend that fisheries be managed to take some fraction of the maximum sustained yield, termed the allowable yield.

In the net productivity model, the yield obtained from Ryder's morphoedaphic index model is taken as an estimate of the MSY for a fish community. However, in order to use the model for a particular fish species, such as lake trout, the MSY for that species must be estimated. This can be done by analyzing the catch statistics, by species, for fisheries which appear to be withstanding current fishing pressures. Recent work by the Ontario Ministry of Natural Resources has produced estimated MSY values for smallmouth bass, whitefish and northern pike, as well as revised MSY values for lake trout and walleye. The MSY value for lake trout, for example, is 0.25 MSY_c, reflecting the fact that lake trout comprised, on average, one-quarter of the total fish catch.

Mathematical formulations for the net productivity model appear in the Fisheries report, where the derivation of the model is described in detail. In addition, the net productivity model is discussed in the Integration report in the context of the Lakeshore Capacity Study as a whole.

8. WILDLIFE COMPONENT

A wildlife community has a network of interrelationships among all species. The organisms are linked together by their effects on one another and their responses to the environment they share (Figure 11).

Shorelines provide a unique set of biophysical environmental features which significantly affect wildlife species. For study purposes, the shoreline communities were defined by the Wildlife Component as those within 100m of the water. The objective was to determine the impact of lakeshore cottage development on wildlife.*

From the wildlife community, the wildlife ecologists selected the following species or groups of species for study: songbirds, common loons, hawks, small mammals, mink, deer, reptiles and amphibians. These were chosen because they represented major parts of the wildlife community, reflected a wide range of values people place on wildlife, and occurred in sufficient numbers to enable investigation.

* This chapter is based on the report Lakeshore Capacity Study, Wildlife (Euler, 1983).

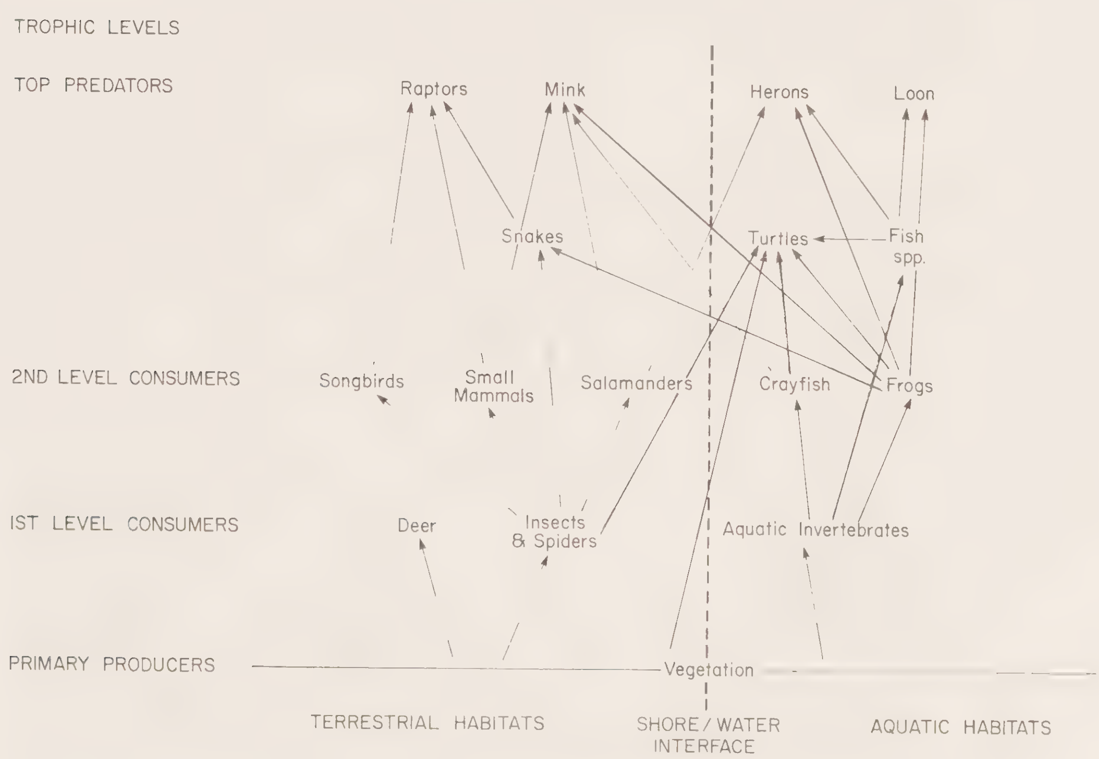

Figure 11. Diagrammatic representation of an ecological food web

Cottage development involves the construction of both cottages and associated structures such as roads, transmission lines, boathouses, beaches and docks. Usually cottagers clear portions of the shoreline for lawns and patios. In addition, they may remove vegetation in the littoral zone to facilitate boating and swimming. An underlying assumption was that the impact of cottage development on wildlife species was caused largely by changes in vegetation. The starting point, therefore, was vegetation measurement and analysis.

Several indices were developed by the wildlife researchers to provide a measure of the vegetation disturbance associated with cottage development (Table 6).

Table 6. Development indices used in the Lakeshore Capacity Study wildlife analysis

	Name	Definition	Application (Wildlife Technical Report Chapter)
ADI	Area Development Index	ADI = GDI + SDI + TDI	Mink (8) Small Mammals (9) Habitat Disturbance (2) Vegetation (1)
GDI	Ground Development Index	Proportion of total area disturbed in ground vegetation layer	Songbirds (7) Habitat Disturbance (2)
SDI	Shrub Development Index	Proportion of total area disturbed in shrub vegetation layer	Habitat Disturbance (2)
TDI	Tree Development Index	Proportion of total area disturbed in tree vegetation layer	Habitat Disturbance (2)
CLDI	Cottager Loading Development Index	CLDI = log(CxU/AxD)	Mink (8)
LDI 150	Loon Development Index 150	Number of cottages within 150m of nest	Loons (4)
LDI 250	Loon Development Index 250	Number of cottages within 250m of nest	Loons (4)

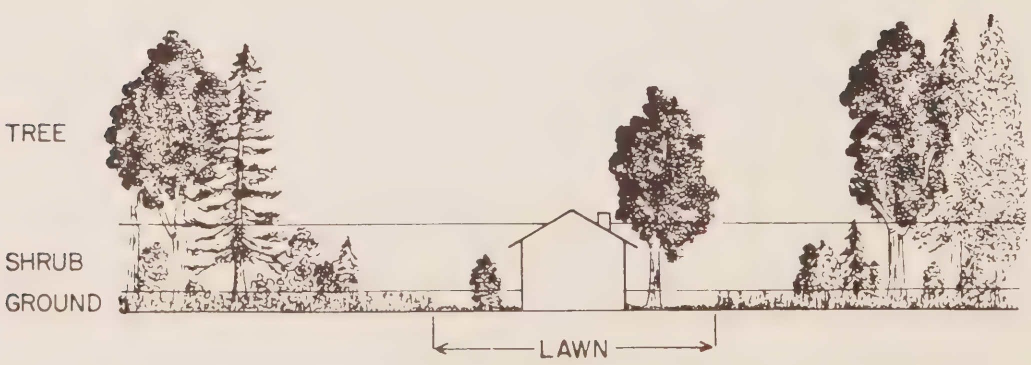

Figure 12. Schematic diagram of a cottage lot, illustrating differential disturbance in the ground, shrub and tree layers

The Area Development Index (ADI) was designed to quantify the disturbance in each of the three vegetation layers, i.e. the tree, shrub and ground layers (Figure 12). The Ground Development Index (GDI) measured the area disturbed in the ground layer. The ADI was used in the analysis of the impact on small mammals and mink; the GDI was used for the songbird research.

An analysis of cottage lots indicated that the area disturbed on a cottage lot was relatively uniform, regardless of lot size. A relationship between the ADI and lot size was identified, making it possible to predict the ADI from information on lot size. Moreover, as the response of many wildlife species to development was measured against the ADI, it was possible to calculate animal responses to cottage development from data on lot size.

To determine how the alterations in vegetation affected wildlife species, it was necessary to define the habitat of each species. Vegetation was sampled in the vicinity of wildlife or of physical signs of wildlife, such as deer tracks or mink dens. The habitat characteristics selected and the experimental design for the study of each species or group of species was based on knowledge of the natural history of the species being examined.

It was assumed that the impact on a particular species was related to the amount of its preferred habitat around the lake. The relationships derived from the research allowed predictions of impact to be made from readily available information on shoreline forest cover type (coniferous, deciduous or mixed wood) and cottage lot size.

The wildlife research indicated that lakeshore lots with cottages had less shrub cover and a lower ratio of coniferous trees. It was also evident that wildlife species differed in their ability to tolerate cottage development; some species could adapt to changes in their habitat while others were sensitive to minor alterations in vegetative composition.

8.1 VEGETATION (WILDLIFE HABITAT)

The vegetation on undisturbed shorelines was analyzed in terms of three types of habitat (coniferous, deciduous and mixed wood) and three layers of vegetation (tree, shrub and ground). Coniferous habitat was found to be typical of the lakeshore for most lakes in the Study Area, while deciduous and mixed habitat types more closely resembled backshore areas.

Data were also collected to provide detailed information about the changes in vegetation associated with development. The two major impacts identified were the replacement of natural lakeshore species by either domesticated or weed species and the overall loss of vegetation. Selected cutting of trees, for example, often resulted in the removal of conifers and the retention of white birch.

8.2 SONGBIRDS

Songbirds depend on the availability of suitable habitat; some species require specific features while others can adapt to a wide range of habitat characteristics. The red-eyed vireo, for example, needs relatively mature trees whereas the American robin can exploit conditions varying from lawns to less disturbed forest.

The reaction of individual species to cottage development was measured, as well as the response of the songbird community. Some species could not tolerate even low levels of development; others seemed virtually unaffected by cottages;

and some even benefited from disturbance. Analysis of the songbird community indicated that extensive cottage development changed its composition (Figures 13 and 14).

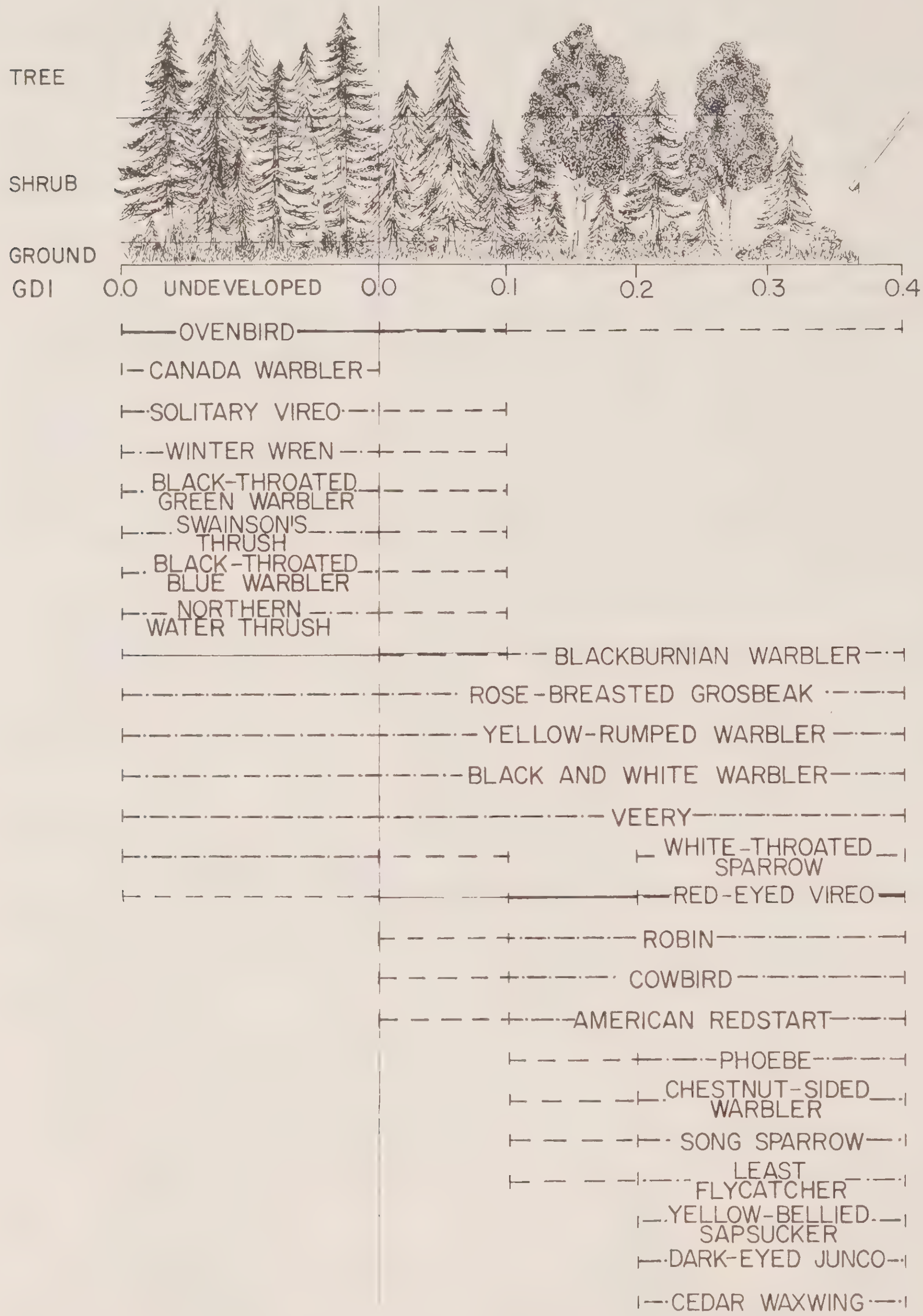

Figure 13. Songbirds in coniferous habitat at varying levels of development

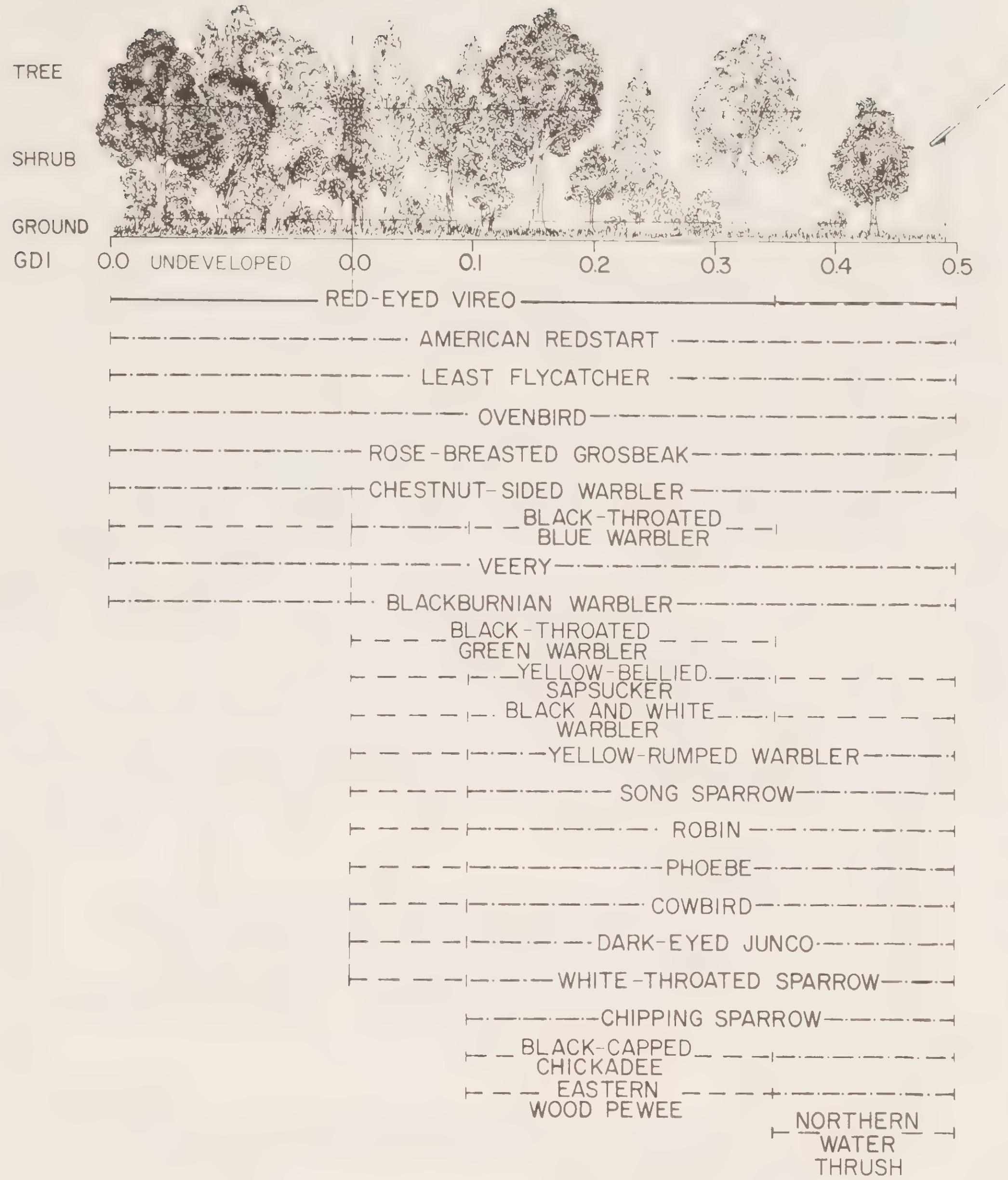

Figure 14. Songbirds in deciduous habitat at varying levels of development

8.3 COMMON LOONS

Nesting sites of the common loon were found primarily on small islands or bog hummocks. Characteristic features included gently sloping shoreline and abundant ground and shrub cover. There was a marked tendency for loons to utilize the same nesting site over a period of consecutive years.

Habitat characteristics were statistically similar for successful (with at least a single egg hatching) and unsuccessful nesting sites. It was assumed, therefore,

that non-habitat factors were relevant. As significantly fewer nests were found in the vicinity of cottages, the level of human activity was thought to be a factor in reduced nesting success.

Loons can live up to 20 years in the wild, so the effects of changing reproductive success cannot be detected in a short-term study. While loons are still commonly observed in the Muskoka-Haliburton area, it is not known whether the loon population is increasing or decreasing.

8.4 HAWKS

The two species of hawks found in high densities in the Study Area are the broad-winged hawk and the red-shouldered hawk. These avian predators have a large hunting range.

The broad-winged hawk needs younger deciduous stands which often occur after the construction of cottages and associated structures. For this reason, cottage development initially tends to improve, or at least maintain, the broad-winged hawk habitat. The greatest source of disturbance appeared to be human activity. This species establishes nests during the latter part of May, which coincides with an influx of cottagers around the Victoria Day long weekend. Such activity could cause nest disturbance or desertion, because defense of the nesting site is weakest during the period of territorial establishment.

Red-shouldered hawk habitat falls into two broad categories: upland deciduous areas and lowlands along streams and small rivers. This species requires a continuous forest of at least 10 ha. Cottage roads could fracture and isolate valuable habitat along lakes and streams.

8.5 SMALL MAMMALS

The small mammals studied included mice, shrews, voles, chipmunks and squirrels. These animals play an important role in shoreline wildlife communities, by converting plant and insect biomass into animal matter and providing a food source for higher level land predators, such as mink and raptors.

The response of small mammals to cottage development varied with the species and the coniferous composition of the habitat. Some species, such as the woodland jumping mouse and masked shrew, not only declined in number as cottages increased but were extirpated at higher levels of development. Other species, including the red squirrel and eastern chipmunk, were more tolerant of development.

8.6 MINK

Mink live along lakeshores and streams throughout most of Ontario. They require a relatively undisturbed habitat and an adequate food supply to reproduce successfully. Aquatic animals like crayfish and frogs as well as terrestrial animals like mice and shrews make up their diet. Mink are good indicators of environmental perturbation, as they are sensitive to changes in water quality, vegetative composition along the shoreline, and the abundance of their food supply.

Mink activity was greatest on undeveloped shorelines with coniferous and mixed vegetation; deciduous shorelines were used very little, regardless of the level of development. Mink activity decreased with even slight increases in the amount of cottage development (Figure 15). The wildlife scientists attributed this to a reduction in available food, hunting locations and denning sites.

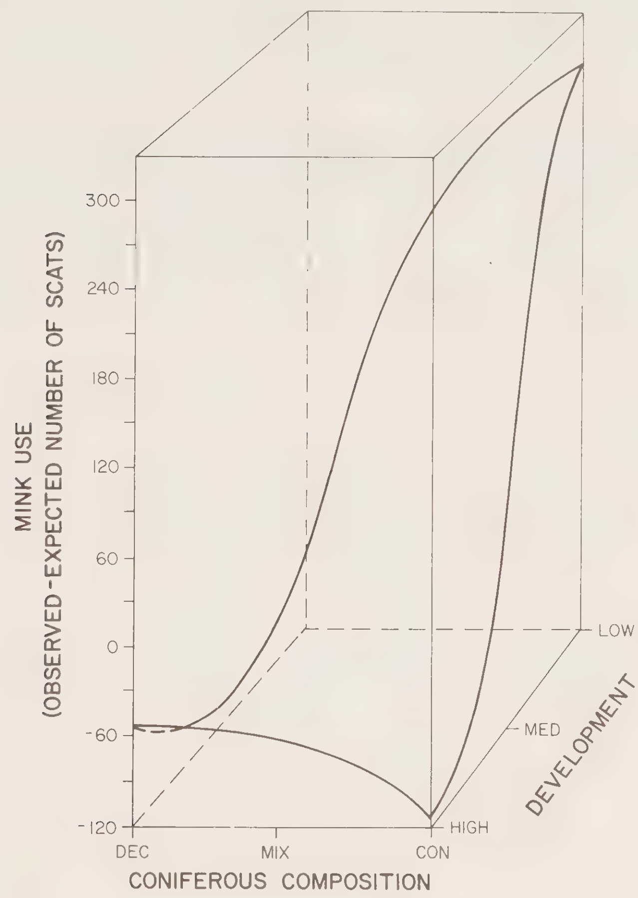

Figure 15. Mink response to development and habitat type

8.7 WHITE-TAILED DEER

Four types of deer habitat were studied, including travel lanes from one conifer stand to another, night bedding sites, day bedding sites, and feeding sites. These habitat types had significantly different characteristics. The night bedding areas and travel lanes had a dense, continuous, closed canopy of coniferous trees to provide protection from severe cold and wind; the day bedding sites had low overhead cover to permit the sun to penetrate; and the feeding sites had the greatest amount of food.

Shorelines with cottages had a lower percentage of coniferous trees and less shrub cover, due to cutting and pruning by cottagers. As tree removal interrupts the continuity of the coniferous fringe, it adversely affects the suitability of the habitat for deer.

8.8 REPTILES AND AMPHIBIANS

Reptiles and amphibians form an integral part of food chains, as predators of insects and their larvae and as prey for higher level consumers.

The most common species of turtle in Muskoka-Haliburton are the snapping turtle and eastern painted turtle. For optimum breeding success, turtles require sandy areas free of vegetation, with a clear path to the water. Turtles live up to 20 years, so it was not possible to determine their long-term response to cottage development. Nevertheless, it was clear that the increased road traffic and destruction of egg-laying sites associated with cottages had a negative effect.

Amphibians are most common in the moist habitat along streams and lakes. As most species require terrestrial and aquatic habitat, it was necessary to examine both the vegetation and water chemistry of their habitat. Densities of amphibians were measured throughout the spring and summer along streams, lakes and ponds. For all amphibians studied except bullfrogs and spotted salamanders, disturbance caused by cottage development was detrimental to habitat quality.

8.9 LAKELIFE

The results of the wildlife research were incorporated into a computerized information system known as LAKELIFE, which predicts the effects of cottage development on wildlife. A full description of LAKELIFE appears in the Lakeshore Capacity Study report on Wildlife.

The predictions are in the form of indices for each species or group of species. Some of these are calculated from estimates of certain vegetation parameters which have been shown to change with development; others depend on the Area Development Index (ADI). The calculations and assumptions used in LAKELIFE are described in the Wildlife report. In addition, the methods of predicting the impact of lakeshore cottage development on wildlife are discussed in the context of the Lakeshore Capacity Study as a whole in the Integration report, in the section on the Wildlife Submodel of the Ontario Lakeshore Capacity Simulation Model.

8.10 RECOMMENDATIONS

The recommendations of the Wildlife Component biologists are designed to protect the wildlife community as a whole. Along a stream, for example, the habitat left undisturbed could preserve amphibians and other small aquatic life as well as broad-winged hawk nesting sites, small mammal and songbird habitat, and other wildlife species in the community.

Where minimum habitat requirements are not met, the potential exists for serious losses to individual lakeshore wildlife communities. Clearings for cottage development alter habitat structure sufficiently that the small mammal and songbird species which require a closed canopy, small openings, and/or mature forest would disappear from the shoreline. The combined effect of the reduction in diversity and abundance of songbirds, small mammals, and amphibians would change the natural food web, thereby reducing the number of predators such as mink. Even though these predators could feed on prey species like chipmunks and robins, that increase in number with development, they would still be affected adversely by disruption of their denning and breeding sites.

The Wildlife Component biologists made specific recommendations with respect to the degree of acceptable change in wildlife communities as a result of cottage development. As decisions regarding acceptability involve subjective judgement and reflect attitudes and beliefs, the wildlife biologists may not always be in full agreement with planners or decision-makers. This does not detract from the value of the wildlife recommendations, as long as their underlying premises are clear.

Two major premises formed the basis for evaluating the acceptability of changes in the wildlife community:

i. Wildlife populations must be maintained in self-sustaining communities that are as similar as possible to undisturbed shoreline communities, on some portions of all lakes.
ii. There must be no extirpation of any species from the shoreline community of any lake.

The following specific recommendations of the Wildlife Component biologists are designed to benefit the wildlife community as a whole.

(1) A Lakeshore Habitat Monitoring System should be introduced in areas where lakeshore cottage development occurs. This system should:
(a) identify all potentially significant impacts of cottage development on lakeshore wildlife and wildlife habitat for each lake;
(b) build an inventory of lakeshore habitats; and,
(c) act as a data base with which cottage development plans can be evaluated, using LAKELIFE to test different management strategies.

The first stage of this system should involve the identification of sensitive areas on the lake shoreline and the collection of data required to execute the program for the LAKELIFE Simulation Model.

(2) All shoreline habitat for rare, threatened, or endangered species, both plant and animal, should be treated in accordance with current legislation and protected.

(3) Development should not be permitted if wetlands are to be filled or isolated from the lake. A wetland is any area covered by standing water until about July 1st each year.

(4) Any lakeshore identified as a winter deer concentration area should be protected, but the area reserved need not exceed 5% of the total shoreline length. Service lines and roads passing through these areas to cottages elsewhere on the lake should be at least 120m from the shoreline.

(5) To preserve at least one pair of nesting loons on each lake, two islands of at least 0.5 ha should be left undeveloped. On lakes without two islands of that size, two areas of at least 500m of shoreline and 100m inland should be reserved, each centred on a potential loon nesting site.

(6) To maintain hawk nesting sites along lakeshores, at least two actual or suitable nesting areas should be designated. No cottage development should be permitted within 150m.

(7) At least two potential turtle egg-laying sites should be left undeveloped on each lake. Each site should be at least 1 ha in area, with at least 100m of shore.

(8) To maintain amphibian populations, 50% of the segments (as defined in LAKELIFE) containing a stream entering or leaving a lake should remain undeveloped.

(9) The model LAKELIFE should be used to predict the impact on wildlife habitat of proposed cottage developments. It reduces the complex requirements of all these species to a series of simple values which can be easily interpreted. The minimum levels of values computed by LAKELIFE which will allow the premises to be met are the following:

- The small mammal Capture Index should not be below 25% of its original value for each of the seven indicator species.
- Mink activity should not be less than one-third of its value before development. The Cottager Loading Development Index, which reflects good mink habitat, should be below 1.0.
- For each of the 19 songbird species listed in LAKELIFE, 25% of the original nesting habitat should be preserved. In addition, the Coefficient of Community, a measure of the change in bird species, should remain the same on 25% of the lakeshore.

In addition, loon, hawk, turtle and amphibian habitat requirements are incorporated into LAKELIFE, based on the above recommendations.

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APPENDICES

A. LAKESHORE CAPACITY STUDY:
PEER REVIEW

B. LAKESHORE CAPACITY STUDY:
PUBLISHED REPORTS

C. LAKESHORE CAPACITY STUDY:
TECHNICAL PUBLICATIONS

APPENDIX A. PEER REVIEW

To ensure the soundness of the research undertaken by each of the components, the Steering Committee invoked a peer review. For each component, several reviewers were chosen who had practical experience and recognized qualifications in the appropriate field of investigation. Other criteria included specialized knowledge of statistical techniques and familiarity with the Study Area.

Some of the peers have dual roles as university professors and consultants and many are widely known among their professional colleagues, both nationally and internationally.

The names of the peer reviewers are listed, by component. Their affiliations are those at the time of the Lakeshore Capacity Study peer review.

LAND USE COMPONENT	PEER REVIEW
John R. Bousfield, M.C.I.P. John Bousfield Associates Consulting Town Planners Toronto, Ontario	1977
Thomas L. Burton, Ph.D., M.C.I.P. Professor and Chairman Department of Recreation Administration The University of Alberta Edmonton, Alberta	1983
R. S. Dorney, Ph.D., M.C.I.P. Consulting Ecologist Ecoplans Limited Waterloo, Ontario and Professor School of Planning Faculty of Environmental Studies University of Waterloo Waterloo, Ontario	1977 and 1982
Robert W. McCabe, Ph.D., M.C.I.P. Professor Emeritus University of Toronto Toronto, Ontario	1982
J. Ross Raymond, B.A.Sc., P. Eng., M.C.I.P. Planning Consultant Gravenhurst, Ontario	1977

J. C. Stansbury, B.L.A. Hough, Stansbury and Associates Limited Toronto, Ontario	1977
J. B. R. Whitney, Ph.D. EcoPact Assessments Toronto, Ontario and Professor Department of Geography University of Toronto Toronto, Ontario	1977 and 1982

FISHERIES COMPONENT

F. E. J. Fry, Ph.D. James F. MacLaren Limited North York, Ontario	1977 and 1981
R. H. Green, Ph.D. Associate Professor of Zoology Department of Zoology The University of Western Ontario London, Ontario	1981
Harold H. Harvey, Ph.D. Director Aquatic Environmental Affairs Branch Environment Canada Ottawa, Ontario and Professor Department of Zoology University of Toronto Toronto, Ontario	1977
John R. M. Kelso, Ph.D. Great Lakes Biolimnology Laboratory Environment Canada Sault Ste. Marie, Ontario	1977
Henry A. Regier, Ph.D. Institute of Environmental Studies University of Toronto Toronto, Ontario	1981
Carl J. Walters, Ph.D. Associate Professor Institute of Animal Resource Ecology The University of British Columbia Vancouver, B.C.	1981

MICROBIOLOGY COMPONENT

Victor J. Cabelli, Ph.D. Professor Department of Microbiology University of Rhode Island North Kingston, Rhode Island	1977 and 1981
Alfred P. Dufour, Ph.D. United States Environmental Protection Agency Wakefield, Rhode Island and Microbiology Laboratory University of Rhode Island North Kingston, Rhode Island	1981
B. J. Dutka, M.Sc. Head, Microbiology Laboratories Analytical Methods National Water Research Institute Canada Centre for Inland Waters Environment Canada Burlington, Ontario	1977 and 1981
Alfred W. Hoadley, Ph.D. Professor School of Civil Engineering and School of Biology Georgia Institute of Technology Atlanta, Georgia	1977
Patricia L. Seyfried, Ph.D. Associate Professor Department of Microbiology and Parasitology Faculty of Medicine University of Toronto Toronto, Ontario	1981
Alan D. Tennant, Ph.D. Manager, Laboratory Services Environmental Protection Services Ontario Region Environment Canada Ottawa, Ontario	1977

TROPHIC STATUS COMPONENT	
Joseph Shapiro, Ph.D. Professor Geology and Ecology Departments and Associate Director Limnological Research Center University of Minnesota Minneapolis, Minnesota	1977
F. H. Rigler, Ph.D. Chairman Department of Biology McGill University Montreal, Quebec	1977
K. Patalas, Ph.D. Research Scientist Eutrophication Section Fisheries Research Board of Canada Freshwater Institute Environment Canada Winnipeg, Manitoba	1977

WILDLIFE COMPONENT	
J. E. Bryant, M.A.* Director Ontario Region Canadian Wildlife Service Environment Canada Ottawa, Ontario	1981
F. F. Gilbert, Ph.D. Associate Professor College of Biological Science Department of Zoology University of Guelph Guelph, Ontario	1977 and 1980
John A. Livingston, Ph.D. Professor Faculty of Environmental Studies York University North York, Ontario	1977 and 1980
W. John Richardson, Ph.D.** Vice-President, Research LGL Limited Environmental Research Associates Toronto, Ontario	1980

*Review by staff of the Ontario Region, Canadian Wildlife Service.
**Review by staff of LGL, including W. John Richardson, Ph.D., R. A. Davis, Ph.D., and J. Green, M.Sc.

APPENDIX B. PUBLISHED REPORTS

Report	Ministry*
Lakeshore Capacity Study Committee Report**	Municipal Affairs and Housing
Lakeshore Capacity Study Integration	Municipal Affairs and Housing
Lakeshore Capacity Study Land Use	Municipal Affairs and Housing
Lakeshore Capacity Study Trophic Status	The Environment
Lakeshore Capacity Study Microbiology	The Environment
Lakeshore Capacity Study Fisheries	Natural Resources
Lakeshore Capacity Study Wildlife	Natural Resources

*The reports were prepared in the ministries noted. They were published by the Ministry of Municipal Affairs and Housing (1983) and the Ministry of Municipal Affairs (1986).
**A general summary of the Study results.

APPENDIX C. TECHNICAL PUBLICATIONS

TROPHIC STATUS COMPONENT

SCIENTIFIC PUBLICATIONS*

Dillon, P. J., D. S. Jeffries, W. Snyder, R. Reid, N. D. Yan, D. Evans, J. Moss and W. A. Scheider. 1978. *Acidic precipitation in south-central Ontario: recent observations.* **J. Fish. Res. Board Can.** 35:809-815.

Dillon, P. J., and R. A. Reid. 1981. *The input of biologically available phosphorus by precipitation to Precambrian lakes.* pp. 183-198. In **Atmospheric Input of Pollutants to Natural Waters,** S. Eisenreich, (ed.), Ann Arbor Science.

Dillon, P. J., and R. D. Evans. 1982. *Whole-lake lead burdens in sediments of lakes in southern Ontario, Canada.* **Hydrobiol.** 91:121-130.

Evans, H. E., P. J. Smith and P. J. Dillon. 1983. *Anthropogenic zinc and cadmium burdens in sediments of selected southern Ontario lakes.* **Can. J. Fish. Aquat. Sci.** 40:570-579.

Evans, R. D., and P. J. Dillon. 1982. *Historical changes in anthropogenic lead fallout in southern Ontario, Canada.* **Hydrobiol.** 91:131-137.

Jeffries, D. S., W. R. Snyder, W. A. Scheider and M. Kirby. 1978. *Small-scale variations in precipitation loading near Dorset, Ontario.* **Wat. Pollut. Res. Can.** 13:73-84.

Kurtz, J., and W. A. Scheider. 1981. *An analysis of acidic precipitation in southcentral Ontario using air parcel trajectories.* **Atm. Environ.** 15:1111-1116.

Scheider, W. A., W. R. Snyder and B. Clark. 1979. *Deposition of nutrients and ions by precipitation in south-central Ontario.* **Wat. Air Soil Pollut.** 12:171-185.

Scheider, W. A., D. S. Jeffries and P. J. Dillon. 1981. *Bulk deposition in the Sudbury and Muskoka Haliburton areas of Ontario during the shutdown of Inco Ltd., in Sudbury.* **Atm. Environ.** 15:945-956.

MOE DATA REPORT SERIES

Scheider, W. A., R. A. Reid, B. A. Locke and L. D. Scott. 1983. **Studies of lakes and watersheds in Muskoka-Haliburton, Ontario: Methodology (1976-1982).** Ont. Min. Envir. Data Report 83/1.

Jeffries, D. S., and W. R. Snyder. 1983. **Geology and geochemistry of the Muskoka-Haliburton Study Area.** Ont. Min. Envir. Data Report 83/2.

Nicolls, A., R. Reid and R. Girard. 1983. **Morphometry of the Muskoka-Haliburton Study Lakes.** Ont. Min. Envir. Data Report 83/3.

Reid, R. A., B. Locke and R. Girard. 1983. **Temperature profiles on the Muskoka-Haliburton Study Lakes (1976-1982).** Ont. Min. Envir. Data Report 83/4.

Reid, R. A., R. Girard and B. Locke. 1983. **Oxygen profiles on the Muskoka-Haliburton Study Lakes (1976-1982).** Ont. Min. Envir. Data Report 83/5.

Scheider, W. A., C. M. Cox and L. D. Scott, 1983. **Hydrological data for lakes and watersheds in the Muskoka-Haliburton Study Area (1976-1980).** Ont. Min. Envir. Data Report 83/6.

* The analysis of precipitation samples for the Lakeshore Capacity Study in the Muskoka-Haliburton area provided the first scientific evidence of the extent of acidic precipitation in Ontario. These publications are based on Lakeshore Capacity Study data.

Smith, P. J. 1983. **Sediment chemistry of lakes in the Muskoka-Haliburton Study Area.** Ont. Min. Envir. Data Report 83/7.

Nakamoto, L., L. Heintsch and K. Nicholls. 1983. **Phytoplankton of lakes in the Muskoka-Haliburton area.** Ont. Min. Envir. Data Report 83/8.

Hitchen, G. G., and N. D. Yan. **Crustacean zooplankton communities of the Muskoka-Haliburton Study Lakes: Methods and 1976-79 data.** Ont. Min. Envir. Data Report 83/9.

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Dillon, P. J., and W. A. Scheider. **Total phosphorus and major ion mass balances for lakes in the Muskoka-Haliburton Study Area (1976-1980).** Ont. Min. Envir. Data Report

WILDLIFE TECHNICAL REPORT

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1. Clark, T., and D. Euler. 1981. *Vegetation disturbance caused by cottage development in central Ontario.*
2. Racey, G., and D. Euler. 1981. *An index of habitat disturbance for lakeshore cottage development.*
3. Clark, K., D. Euler and E. Armstrong. 1981. *Habitat associations of breeding birds in cottaged and uncottaged areas of central Ontario.*
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5. Armstrong, E., D. Euler and K. Clark. 1981. *Habitat selection within four passerine families in central Ontario.*
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8. Racey, G., and D. Euler. 1981. *Changes in mink habitat and food selection as influenced by cottage development in central Ontario.*
9. Racey, G., and D. Euler. 1981. *Small mammal and habitat response to shoreline cottage development in central Ontario.*
10. Armstrong, E., G. Racey and D. Euler. 1981. *Deer habitat quality and influence of cottage development in central Ontario.*
11. Clark, K., and D. Euler. 1981. *Reptile and amphibian habitat and human activity in central Ontario.*
12. Clark, K. and D. Euler. 1981. *The importance of pH and habitat disturbance in amphibian distribution in central Ontario.*


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LAKESHORE CAPACITY STUDY

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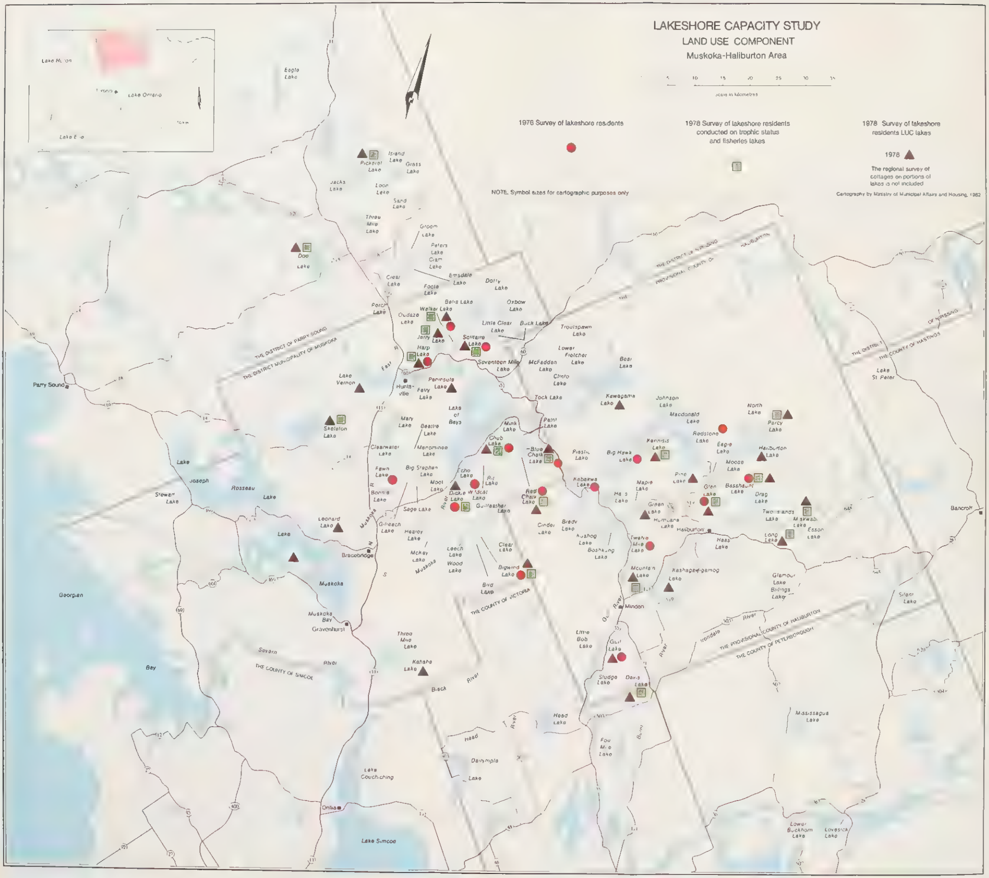
Lower Buckhorn
Lovesick
Mississagua
Lake Couchiching


LAKESHORE CAPACITY STUDY
LAND USE COMPONENT
Muskoka-Haliburton Area
Scale in kilometres
1976 Survey of lakeshore residents
1978 Survey of lakeshore residents conducted on trophic status and fisheries lakes
1978 Survey of lakeshore residents LUC lakes
1978
The regional survey of cottages on portions of lakes is not included
NOTE. Symbol sizes for cartographic purposes only
Cartography by Ministry of Municipal Affairs and Housing, 1982
Lake Ontario
Eagle Lake
Pickerel Lake
Island Lake
Grass Lake
Jacks Lake
Loon Lake
Sand Lake
Three Mile Lake
Groom Lake
Doe Lake
Peters Lake
Cram Lake
Clear Lake
Emsdale Lake
Dotty Lake
Foote Lake
Porch Lake
Bella Lake
Oxbow Lake
Walker Lake
Oudaze Lake
Little Clear Lake
Buck Lake
Troutspawn Lake
Jerry Lake
Solitaire Lake
Harp Lake
Lower Fletcher Lake
Seventeen Mile Lake
McFadden Lake
Bear Lake
Clinto Lake
THE DISTRICT OF PARRY SOUND
THE DISTRICT MUNICIPALITY OF MUSKOKA
THE DISTRICT OF NIPISSING
PROVISIONAL COUNTY OF HALIBURTON
THE COUNTY OF HASTINGS
Lake St Peter
Parry Sound
Lake Vernon
Huntsville
Peninsula Lake
Fairy Lake
Tock Lake
Kawagama Lake
Johnson Lake
North Lake
Percy Lake
Lake of Bays
Mary Lake
Beattie Lake
Mink Lake
Paint Lake
Macdonald Lake
Redstone Lake
Skeleton Lake
Chub Lake
Blue Chalk Lake
Plastic Lake
Big Hawk Lake
Kennisis Lake
Eagle Lake
Haliburton Lake
Clearwater Lake
Menominee Lake
Moose Lake
Fawn Lake
Big Stephen Lake
Echo Lake
Pine Lake
Bonnie Lake
Moot Lake
Kabakwa Lake
Maple Lake
Glen Lake
Basshaunt Lake
Drag Lake
Lake Joseph
Rosseau Lake
Stewart Lake
Dickie Lake
Wildcat Lake
Red Chalk Lake
Sage Lake
Guilfasher Lake
Green Lake
Twelve Mile Lake
Bancroft
Leonard Lake
Gilleach Lake
Cinder Lake
Brady Lake
Kushog Lake
Haliburton
Esson Lake
Long Lake
Healey Lake
Bracebridge
Leech Lake
Clear Lake
Boshkung Lake
Haas Lake
McKay Lake
Wood Lake
Bigwind Lake
Mountain Lake
Kashagawigamog Lake
Glamour Lake
Billings Lake
Muskoka
Georgian Bay
Bird Lake
THE COUNTY OF VICTORIA
Minden
Silent Lake
Muskoka Bay
Gravenhurst
Three Mile Lake
Little Bob Lake
Gull Lake
Irondale River
THE PROVISIONAL COUNTY OF HALIBURTON
THE COUNTY OF PETERBOROUGH
Severn River
THE COUNTY OF SIMCOE
Kahshe Lake
Black River
Sludge Lake
Davis Lake
Head Lake
Mississagua Lake
Four Mile Lake
Burnt River
Head River
Dalrymple Lake
Lake Couchiching
Orillia
Lake Simcoe
Lower Buckhorn Lake
Lovesick Lake

FOREWORD

Community planners and other professionals involved in the preparation of planning policies for lakeshore development and in the review of specific subdivision proposals have always found it difficult to determine objectively the impact of development on the natural environment. In response to this challenge, the Lakeshore Capacity Study was undertaken to provide a planning tool to assist in evaluating the effects of cottage development on inland lakes and lakeshores. Central to the task was the need to gain a clearer understanding of the relationship between cottage development and its impacts on selected aspects of the natural environment.

To accomplish these objectives, the Ministry of Municipal Affairs carried out the Lakeshore Capacity Study in cooperation with the Ministry of the Environment and the Ministry of Natural Resources.

The Muskoka-Haliburton area of central Ontario was chosen as the Study Area. The homogeneity of the area, which is part of one physiographic region, reduced the need to account for major natural variations among the lakes and watersheds. In addition, the extent of existing development on the lakes varied, permitting an examination of situations ranging from no development to full development.

The Study involved measuring the source of environmental impact, in terms of the lakeshore cottages and their use, and the impact of cottage development in terms of water quality, fisheries, and wildlife habitat. The methods of prediction derived from the research were linked in a simulation model, which is capable of predicting trends in the values of the key indicators of impact.

As the research, analysis and findings of the Study are documented in a set of seven reports, selective reading may be desirable. Those readers who have a general interest in the work are advised to read the Committee Report first, as it provides an overall summary of the findings. This should be followed by the Integration report, in which the simulation model is described. Readers with more specialized interests will find the details of each component of the Study in the other five reports. In each, the reader can select from the table of contents the most important chapters for his or her purposes.

The end product of the Study, the Ontario Lakeshore Capacity Simulation Model, has several features worth noting. The spatial unit addressed by the model is a single lake and the lakeshore. When the model is applied, the number of unknowns related to the natural environment can be substantially reduced, making it easier for planners or other professionals to weigh the environmental effects of development. The model goes a step further to permit predictions of the impact of cottage development when different management policies are selected.

The scope of the simulation model demands some explanation. In its present form, the model applies to cottage development on inland lakes in the Study Area, where the research was conducted. However, the methods of prediction can be adapted to other parts of the province, as long as differences in conditions are taken into account.

The purpose of the Study was to measure the environmental impact of cottages. Commercial and industrial uses were excluded deliberately, in order to simplify the difficult task of measuring cottage impact. The flexibility inherent in the simulation model makes it possible to add other types of land use later, if so desired.

Most of the existing cottage development in the Study Area is located in a single tier along the shoreline. For this reason, the simulation model applies to the immediate lakeshore and not to backshore development. Again, the methods of prediction developed for cottages near the lake can be adapted to measure the impact of cottage development in other forms.

The model was designed to measure the physical and chemical impacts of cottages. Accordingly, it does not address other planning concerns, such as social and economic impacts. While these were recognized as essential considerations in decision-making, the specific objective of Phase III of the Lakeshore Capacity Study was to find practical ways of producing better technical data regarding environmental impact.

Now that Phase III of the Study is completed, with the production of the Ontario Lakeshore Capacity Simulation Model (OLCSM), the next step envisaged is to apply the model experimentally within the Ministry. In this setting, model output can be tested in a variety of actual development situations. When this period of experimental use has been concluded and the results assessed, the three participating ministries will be able to determine whether the model should be adapted to other parts of the province and whether it should be made available more widely.

The Ministry of Municipal Affairs considers the OLCSM to be a potential planning tool but recognizes that the technical and organizational implications of its use must be examined. While this is underway, the model will be available for testing as an additional planning tool to supplement the information normally required to evaluate a planning policy or development proposal. However, the model will not be used in the decision-making process, which will still rest on the customary range of planning considerations.

This Land Use report describes the models developed by the Land Use Component planners and social scientists to predict cottage use.

ACKNOWLEDGEMENTS

The Land Use Component predictive models could not have been developed without the cooperation of cottagers in the Muskoka-Haliburton area who responded to our questionnaires. Municipal and regional government staff in the Study Area also assisted, by supplying information regarding road location, quality and maintenance.

Three senior research analysts with the Land Use Component, Laurier F. Therrien, Nicholas J. O. Miles and William K. Carroll, made major contributions at crucial stages of the work.

Dr. Peter H. Peskun, statistical consultant to the Land Use Component, provided penetrating insights regarding statistical possibilities and implications. His suggestion to introduce benchmarking made it possible to convert the Cottager Model from a theoretical to a practical predictive tool.

In addition, valuable suggestions emanating from the peer review assisted the author in the final editing of this report.

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Manager
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At the request of the Lakeshore Capacity Steering Committee, the following specialists reviewed the Land Use Component work program (1977) and this report (1982). Their contributions are acknowledged with thanks.

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PREFACE

This report is intended to provide a general understanding of the two models developed by the Land Use Component research group to predict cottage use. Selected supporting data are included.

Two technical reports provide supplementary material. The Technical Report on Cottage Typology (Miles, 1980) describes the analysis of cottage use patterns. The Technical Report on Land Use Component Models to Predict Cottage Use (Carroll et al., 1982) documents the modelling process in more detail.

The Land Use Component models are included in the Ontario Lakeshore Capacity Simulation Model (OLCSM), which is described in the report on Integration (Teleki and Herskowitz, 1986). Forecasts of cottage use generated by the Land Use Submodel act as the driving variables in the OLCSM, which predicts the impact of cottage development on water quality, fisheries and wildlife.

Step-by-step instructions for applying the Land Use Submodel appear in the OLCSM User's Guide (Teleki and Herskowitz, 1982).

1. INTRODUCTION

The Province of Ontario contains thousands of inland lakes which offer a wide variety of recreational opportunities. The lakes are popular for swimming, boating and fishing during the summer and for downhill skiing, cross-country skiing and snowmobiling during the winter.

For many years, community planners in Ontario have had difficulty evaluating subdivision proposals for cottage development on the shores of inland lakes. The problem stems from the absence of technical knowledge with which to determine objectively whether the cottages are likely to have adverse effects on the lake and surrounding land.

Similar problems are encountered in drafting official plan policies for cottage areas. Even more specific aspects of lakeshore planning, such as zoning standards and site planning, are hampered by the lack of relevant technical data.

Without a recognized method of predicting impact, conflicting claims regarding the ecological repercussions of lakeshore cottages tend to generate confrontation.

While development proponents sometimes minimize the adverse effects, cottage owners often foresee dire consequences. Reconciliation of such extreme positions is unlikely, as long as crucial gaps remain in scientific knowledge of the ecological relationships inherent in lake-watershed systems.

1.1 LAKESHORE CAPACITY STUDY

The objective of the Lakeshore Capacity Study Phase III was to develop a systematic method of predicting the impact of lakeshore cottages on selected aspects of the inland lake environment. It was clear from the initial work in Phases I and II that an in-depth examination of certain specific lake features, such as water quality and fisheries, could be expected to produce practical results.

Figure 1. Muskoka-Haliburton Study Area

Further, the land elements of the land-lake system could be taken into account through wildlife-habitat investigations. Related to each of these was the need for more effective ways of quantifying cottages on the lakeshore, to enable identification of consistent relationships between cottages and water quality, fisheries and wildlife.

As current scientific understanding of lake-watershed systems is incomplete, the model to be developed in Phase III was not viewed as the ultimate solution. Rather, a more realistic expectation was that it would be sound enough to merit practical application and flexible enough to allow future refinement.

Finally, it was clear that no model could solve all lakeshore planning problems. Nevertheless, by measuring the probable impacts of cottage development, the model could substantially reduce the number of unknowns with respect to environmental impact, making it easier for a planner to weigh the environmental factors along with other considerations. In this way, the model could assist in lakeshore planning at the provincial, regional and local levels.

1.2 STUDY AREA

The Study Area selected was the Muskoka-Haliburton area of central Ontario. As this area was within one physiographic region on the Precambrian Shield, it had similar soils and plant communities, thus reducing the need to account for major natural variations among the lakes and watersheds.

With respect to cottages, the extent of existing lakeshore development varied, permitting examination of situations ranging from no development to full development. The largest urban complex within a reasonable driving distance was Metropolitan Toronto, with a population exceeding two million (Figure 1).

1.3 ROLE OF THE LAND USE COMPONENT

The Land Use Component was one of several components responsible for the research necessary to develop models capable of predicting the impact of lakeshore development on the environment (Figure 2).

The specific role of the Land Use Component was to investigate the lakeshore cottages, with a view to deriving a more appropriate method of quantifying cottage development than simply counting the number of cottages. The assumption was that cottages varied in the degree of impact they generated, depending on the type of sewage disposal system, size of lot and level of cottage use. This report is focussed on cottage use.

2. OBJECTIVE

The objective of the Land Use Component was to develop a model to predict the level of cottage use for lakeshore cottages on a specific lake, as input to other Lakeshore Capacity Study models.

Two units of measurement were chosen to describe cottage use. One was "cottage-days", where one cottage-day represented one cottage in use for one day. The other was "user-days", where one user-day represented one cottage in use for one day by one person. Thus, the cottage use for a family of four people, at the cottage for three days, would be recorded as three cottage-days or twelve user-days.

Some of the implications of the objective merit elaboration. As the cottage use predictions would be needed prior to approval of a development proposal, it would not be possible to draw on data regarding cottages or cottagers to assist in forecasting. Thus, the timing would place some constraints on the selection of possible predictors.

Another consideration was the need for compatible units of measurement, as the cottage use predictions were intended as input to other Lakeshore Capacity Study models. The Trophic Status model, for example, was being designed to predict the phosphorus concentration in the lake. As the amount of anthropogenic phosphorus contributed to the lake via septic systems was related to the number of people at the cottages during the year, the appropriate cottage use measurement would be user-days (annual). In contrast, the Fisheries net productivity model was being designed to predict fish harvest. This was dependent in part on the angling effort by cottagers which, in turn, was related to the number of anglers from cottages on the lake. As the latter could be derived from the number of days cottages were occupied during the fishing seasons, the appropriate cottage use measurement would be cottage-days (during fishing seasons).

Further, a practical model must be able to produce cottage use projections expeditiously. Accordingly, the variables selected as predictors would have to be restricted to those for which values would be readily available, so input data could be assembled quickly.

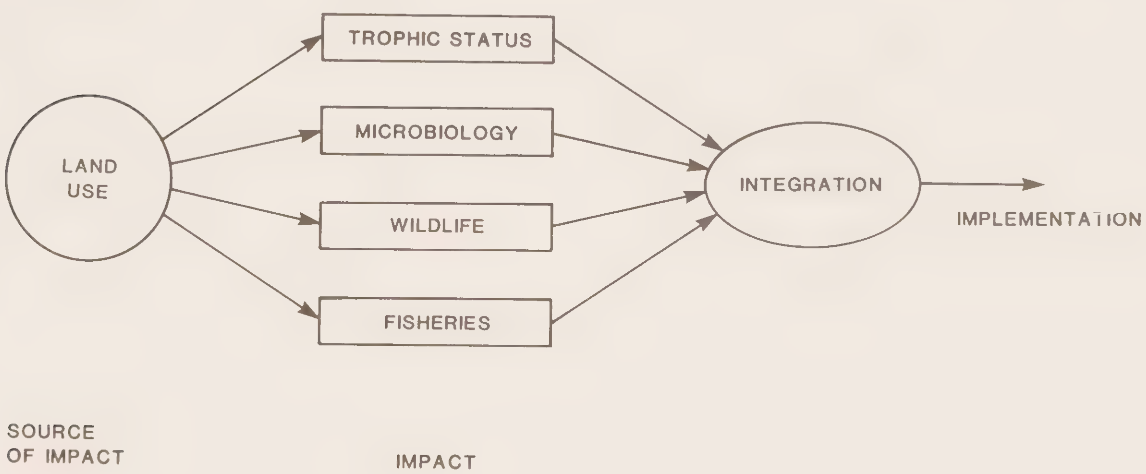


Figure 2. Components of Lakeshore Capacity Study

3. LITERATURE REVIEW

A search of the literature, in 1977, revealed that considerable work had been done on the use of cottages but much of it was descriptive rather than quantitative. Even in the latter category, while many studies produced estimates of annual cottage use, few identified the factors associated with different levels of use. The relevant material noted in this chapter pertains to the level of cottage use, existing models to explain variation in cottage use, and planning policies influencing cottage use.

3.1 LEVELS OF COTTAGE USE

The number of days cottages were occupied during the year was recorded in various studies, as a measure of cottage use. Selected examples are recorded here, for comparison with the findings of the Land Use Component for lakeshore cottages in the Muskoka-Haliburton Study Area.

A report by the American Society of Planning Officials on the impact of second home development in the United States indicated that the median occupancy during the year was 53 cottage-days (ASPO, 1976). The figure varied by region, from a low of 47 in the West to 59 cottage-days per year in the Northeast. Occupancy also varied by season, depending on such factors as seasonal recreational facilities at the second home and the time of year when families traditionally took their vacations.

In Wisconsin, a study of lake property owners showed that the average use of the property was 118 cottage-days per year (Klessig, 1973). When permanent residents, who comprised 18% of the sample, were excluded, the average use was 60 cottage-days.

In Alberta, a survey of cottages on Lac La Biche (Runge and Steffen, 1977) indicated that each cottage generated an average of 0.54 person-years of use, or 197 user-days, during the period from May to October. Winter use was not included, although 28% of the respondents used their cottages in winter. The method of recording cottage use on a simple calendar, with an open square for each day during the survey period, was subsequently adapted for use in the Land Use Component questionnaire.

In Ontario, the Ministry of Natural Resources employed data from the Ontario Recreation Survey to estimate cottage use (Ross and Peck, 1979). By relating the average use of cottages owned by Ontarians in Ontario to the average party size for those using a cottage, it was estimated that cottages were used about 49 to 57 cottage-days during the year.

A study of recreation and community development in the District of Muskoka and the Town of Bala was conducted in 1967, under a Federal/Provincial Rural Development Agreement (Project Planning Associates, 1970). Data from the survey of cottages indicated that the average annual use of a cottage was 80 cottage-days. Families with higher incomes used their cottages more, for an average of 85 cottage-days.

Cottages in Muskoka were surveyed again in 1972, as part of a study of the role of recreation in the economy of the District of Muskoka (JMT Engineering, 1973). The average cottage use during the year was 72 cottage-days.

Based on these studies, the median cottage use in the United States appears to be in a range similar to the average use of cottages in Ontario. The somewhat higher averages in Muskoka may be due to a higher percentage of permanent residents in the sample.

3.2 EXISTING MODELS TO EXPLAIN VARIATION IN COTTAGE USE

In the published literature, two studies were described which attempted to explain variation in the use of existing cottages.

In the early 1970's, the Upper Great Lakes Regional Commission funded a study of lake property owners in northern Wisconsin (Klessig, 1973). The purpose was to learn more about the lake property owners, in order to determine their potential role in lake management. Step regression analysis was used to determine the reasons for variation in the time spent at the lake property. The three factors identified were the number of improvements on the property, the age of the owner and the location (rural or urban) of the owner's permanent residence. Together, they explained 43% of the variance in the annual number of cottage-days at the lake.

In 1970, the College of Agriculture at Pennsylvania State University undertook a study of environmental quality aspects associated with seasonal home communities (Gamble et al., 1975). Most of the communities studied were located on lakes, with a few on the ocean or a bay. One of the objectives was to determine the attitudes of seasonal home occupants regarding environmental quality. Regression techniques were used to explain the reasons for the variation in annual use of the second home. The most significant variable was the seasons of the year the home was used – for each additional season, the use of the home increased by nine cottage-days. Also, older families and families who were willing to accept

restrictions on their activities used their seasonal homes more. Lower use was associated with excessive noise, poor water quality and less emphasis on the quality of the home and community. Together, these variables explained 26% of the variation in seasonal home use.

Some of the factors identified in these two studies as being associated with different levels of cottage use might assist in estimating the use of existing cottages. However, it was not feasible to employ them in attempting to predict cottage use prior to development, because information on cottages and cottagers would not be available at that stage.

3.3 PLANNING POLICIES INFLUENCING COTTAGE USE

When the conceptual approach to a model to predict cottage use was developed, it was recognized that planning policies and zoning regulations could affect access to the cottage and, in turn, cottage use.

Subsequently, problems associated with a change from seasonal to permanent use of cottages in Ontario were documented in a discussion paper on seasonal residential conversions (Ontario Ministry of Housing, 1978). Among the problems were demands that roads be improved and kept open all year. Winter road maintenance and snow clearing for weekend recreational use was considered inadequate for permanent use. It was clear that the standards were lower for roads providing access to seasonal cottages than for those serving permanent residences.

Some municipalities used "Seasonal Residential" zoning to prohibit permanent use of cottages and thereby limit increases in the cost of services. In the Discussion Paper, an alternative concept of "Limited Services" zoning was advocated, as a means of discouraging permanent use. It was argued that it would be easier to enforce than seasonal occupancy restrictions.

4. CONCEPTUAL APPROACH TO THE ACCESSIBILITY MODEL

Several possible methods of predicting cottage use were considered by the Land Use Component group. One simple method was to assume that each cottage might be used all year, so the number of cottages would simply be multiplied by 365 days to estimate the annual cottage-days. While this "worst-case" assumption would undoubtedly be safe, it would drastically overstate cottage use. This was evident when it was compared with the average annual use of cottages recorded in previous studies. For cottages in Muskoka, estimates of actual use ranged from 72 to 85 cottage-days, which was less than one-quarter of the maximum use (365 cottage-days) during the year.

The implications of high predictions were considered. As the water in the lake is adversely affected by cottage use, the simulation model would respond to high cottage use estimates by forecasting lower water quality. In response to such a forecast, stringent limits would be placed on the creation of cottage lots, in order to preserve environmental quality. If this were to occur on many inland lakes, the effect on the total supply of cottage lots could be immense and the possibility of locating a cottage on the lakeshore could become more and more remote. In view of the implications, the "worst-case" approach was rejected.

4.1 AVERAGING MODEL

Another simple method of predicting was to assume that the average number of cottage-days during the year would apply to both existing and proposed cottages. As this method recognized actual levels of cottage use, the predictions would be realistic when averaged for all lakes. The problem with the Averaging Model was its inability to reflect differences in cottage use between lakes.

4.2 ACCESSIBILITY MODEL

In order to develop a model that would be sensitive to lake-to-lake variations in cottage use, the first step was to recognize the constraints imposed by the need to make predictions prior to cottage construction. At that point in time, no information would be available regarding improvements on the lot; number of seasons the cottage was occupied; family size, age or income; or attitudes and perceptions regarding noise or water quality.

The next step was to identify the kinds of information that would be available when a proposed development was submitted for approval. The major items were location of the subdivision, number of lots, and type of access to the lots. Official plan policies and zoning regulations for cottage areas, if any, would also be known. Each of these was examined in the light of its potential value as a predictor of cottage use.

If the time required to reach the cottage was a factor affecting cottage use, then any aspect of cottage accessibility that would tend to increase or decrease the travel time could be relevant. The underlying assumption was that good access was associated with higher cottage use while difficult access was associated with lower cottage use.

Some of the lakes in the Study Area had a considerable number of small islands. As many cottagers consider a cottage on an island in Muskoka the quintessential cottaging experience, some cottages had water access only. These were unlikely to be used extensively in winter, so it was assumed that cottages accessible only by water would tend to have lower cottage use. Thus, the type of access to the cottage was selected as a possible independent variable, to assist in predicting cottage use.

The climate in the Muskoka-Haliburton area also affected cottage access, as snow-clearing was required to keep roads open during the winter. While major roads were cleared, many of the narrow or unimproved roads leading to cottages were closed. Accordingly, another possible explanatory variable was access road open in winter.

Other variables were sought that would reflect either the degree of difficulty or the length of time involved in reaching the cottage. It was evident from field experience in the Study Area that unimproved roads were often muddy in wet weather, making some cottages temporarily inaccessible. Accordingly, the distance travelled on poor roads to reach the cottage was selected as a third possible predictor.

Another aspect of cottage accessibility was the distance from the cottage to urban centres in the region. The rationale was that permanent residents would need banking, medical and other services that were available only in an urban centre. Therefore, those who intended to use their cottage as a permanent residence would be more likely to choose locations reasonably close to one of the larger urban centres in the Study Area. In contrast, those who used their cottages on weekends or long weekends usually brought their supplies from home, so they might tend to choose more remote locations. Based on this reasoning, the distance from the cottage to the nearest

urban centre was added as a fourth possible explanatory variable.

The method of determining appropriate values for these accessibility variables was an intrinsic part of the conceptual approach. The ideal prediction would indicate the level of cottage use each year, for the life of the cottage. While the ideal was not attainable, it was feasible to use official plans, transportation plans and other planning documents as legitimate sources of information for coding the independent variables. This would ensure that future conditions were taken into account in the model, to the degree possible at the time of the prediction.

This approach could make a considerable difference in the predicted annual cottage use. For example, if a proposed subdivision on the lakeshore had water access only at the time of the proposal, and water access was specified in operating the model, the predicted cottage use would be relatively low. But if the official plan included a proposed major road that would provide access all year to the subdivision, and road access all year was specified in the model, the predicted cottage use would be relatively high. Thus, the recognition of planning documents and the incorporation of planned future conditions was a vital part of the Accessibility Model conceptual approach.

The various steps already described, from the application for subdivision approval to the prediction of the annual number of cottage-days per cottage, are set out in the Accessibility Model conceptual diagram (Figure 3). Knowing the number of lots proposed, it would be possible to estimate the maximum number of cottages, as zoning by-laws usually permit only one principal building on a registered lot.

Where the type of access was the same for each lot, the predicted annual cottage-days for the subdivision would be calculated by multiplying the predicted annual cottage-days per cottage by the number of lots (cottages). Where the type of access differed, lots with the same type of access would be grouped, a prediction would be made for each group, and the separate predictions would then be aggregated to provide the prediction for the entire subdivision.

The final step involved conversion of the predicted cottage-days for the subdivision to user-days. This would be accomplished by multiplying the predicted number of cottage-days for the subdivision by the average number of persons per cottage for the entire sample of cottages surveyed. This "global" average would be used for every subdivision.

This conceptual approach was used as a guide in determining the scope of the cottage use data required from cottagers. It was also a reference point for selecting information needs regarding roads, distances, snow clearing, planning policies and zoning regulations.

To sum up, variations in average annual use between cottages on different lakes would be taken into account in the Accessibility Model by using as predictors selected accessibility characteristics associated with cottage use. Similarly, variations in annual cottage use between cottages on the same lake would be taken into account by making separate predictions for groups of cottages with different types of access.

4.3 COTTAGER MODEL

When the conceptual approach to the Accessibility Model was formulated, the possibility of employing cottage use information to assist in predicting was dismissed because it was not considered feasible to obtain such data prior to cottage construction. Later, when new insights into cottage use patterns were provided by the cottage typology analysis, the possibility of exploiting the data for predictive purposes was explored. The result was the conceptual framework for the Cottager Model. It is described in Chapter 8, following the description of the cottage typology in Chapter 7.

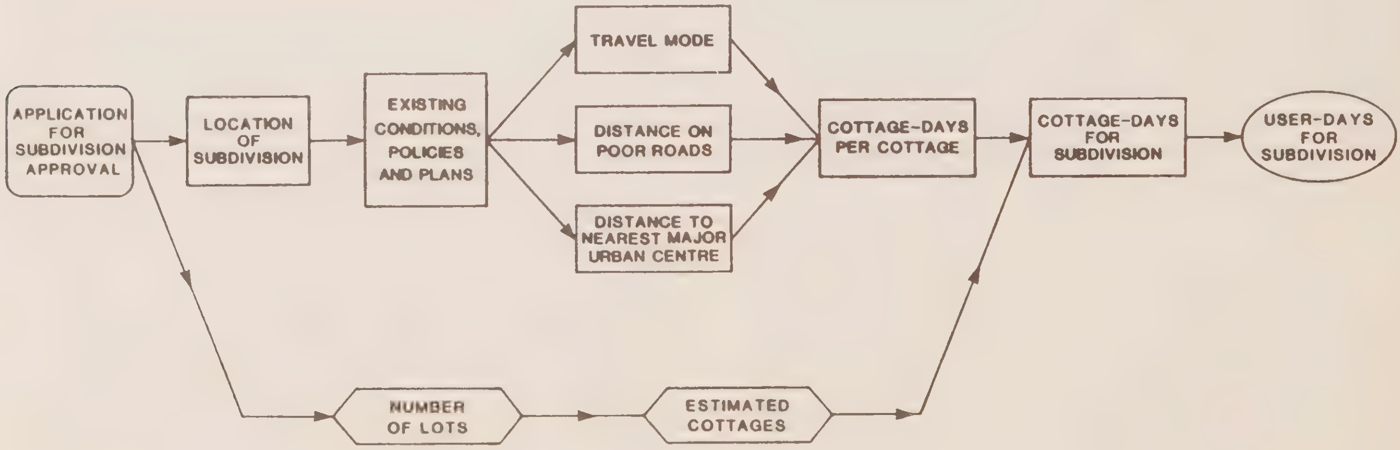


Figure 3. Accessibility Model: a conceptual diagram linking cottage accessibility and cottage use

5. RESEARCH DESIGN

The Land Use Component research to develop a model to predict cottage use was designed to proceed in sequential stages. Each stage depended in part on the preceding work and each contributed to the subsequent work.

The literature search and the conceptual approach have been described in Chapters 3 and 4, respectively. The latter provided a clear focus for selecting the data required for model construction. This included information on cottage use, type of access to the cottage, road standards and distances from the cottages to urban centres in the region. The cottage use data was collected from the cottagers, through a sample survey. Information on roads was obtained from the Ministry of Transportation and Communications, regional and local governments in the Study Area, and field checks. Relevant planning policies and controls were identified through contacts with local, regional and provincial planning departments.

The cottage use information was analyzed before proceeding with the modelling process. Packaged computer programs in the Statistical Package for Social Sciences (SPSS)* were employed to provide data on annual cottage-days, annual user-days, patterns of cottage use throughout the year, and household size, for individual lakes and for the entire sample.

The process of model building was then undertaken. Multiple regression analysis was selected as the best method of developing a linear regression equation. The regression analysis identified the variables with the most influence on the forecast and indicated the weights to be attached to each. The analysis also identified the variables with the least influence on the forecast, which assisted in eliminating unnecessary variables.

The candidate variables measured the various accessibility characteristics described in the conceptual approach to the Accessibility Model. Both continuous and dichotomous variables were employed. The continuous variables, such as distance from the cottage to the nearest urban centre, were expressed as distance measurements. The dichotomous variables were treated as dummy variables.*

To assist in selecting the best set of independent variables, five stepwise regression procedures were employed: forward selection, backward elimination, stepwise, maximum R^2 and minimum R^2.** The conceptual and technical aspects of these procedures are documented in standard statistical text books.

* SPSS User's Guide, 1979 Edition. SPSS Inc., Chicago, Illinois, U.S.A.

At each step, the values of several standard statistical measures were considered. These are described in the Lakeshore Capacity Study Technical Report **Land Use Component Models to Predict Cottage Use** (Carroll et al., 1982). Specifically, they included the coefficient of multiple determination, mean square error, Student's t test and analysis-of-variance F-ratio.

Packaged computer programs in the Statistical Analysis System (SAS) were employed for modelling.*** This system included, in addition to the standard statistical measures, Mallory's Cp and the predicted error sum of squares.

Mallory's Cp assisted in identifying the equation with the least number of independent variables which could predict nearly as accurately as the equation with all appropriate predictors. The Cp value approximates the number of variables in the regression equation when the mean square error approximates the mean square error for the equation including all appropriate predictors.

The predicted error sum of squares assisted in evaluating the predictive power of each regression equation. If the regression equation could predict the dependent variable (cottage-days) with the same degree of accuracy for another sample of cottages drawn from the same area, the value would be similar to the value of the error sum of squares.

The final step was to evaluate the accuracy of the model predictions. This was accomplished by comparing the predictions of cottage use with actual cottage use, for each lake surveyed. The model predictions were based on imputed data and benchmark data; "actual" cottage use was the information obtained from cottagers through the survey.

To sum up, the Land Use Component research to develop a model to predict cottage use was designed to progress through six related stages, namely, literature search; conceptual approach; data collection; data analysis (cottage use); model construction; and evaluation of the accuracy of model predictions. The organization of this report reflects the research design.

* Dummy variables are assigned a value of either 1 or 0, to indicate the presence or absence of a particular condition. In this case, the travel mode variables were treated as dummy variables. Thus, the travel mode variable depicting "road in summer; road in winter" was given a value of 1 if the cottage was accessible by road in summer and in winter; otherwise, it was assigned a value of 0. As the travel modes were mutually exclusive, only one travel mode could have a value of 1.

** Computer symbols are used in this report to denote statistical measures.

*** SAS User's Guide: Statistics, 1979 Edition. SAS Institute, Inc., Cary, North Carolina, U.S.A.

6. DATA COLLECTION

Information regarding cottages and cottagers in the Study Area was collected through two surveys, each of which played a special role in the work of the Land Use Component. The 1978 Survey of Lakeshore Residents concentrated largely on cottage use and the responses provided the data base for developing the predictive models. This major survey and the supplementary survey (Autumn Survey of Lakeshore Residents) which followed it are described in this chapter.

6.1 1978 SURVEY OF LAKESHORE RESIDENTS

The 1978 Survey of Lakeshore Residents was designed to collect data from a representative sample of cottages on inland lakes in the Lakeshore Capacity Study Area, as a basis for model development.

A specific boundary was needed for survey purposes, so the boundaries of the District Municipality of Muskoka and the Provisional County of Haliburton were chosen, with two adjustments. On the west, the Georgian Bay lakeshore was excluded, because the Great Lakes were not considered inland lakes in the context of the Lakeshore Capacity Study. On the east, the townships in Algonquin Park were excluded, because land development in a provincial park did not require approval under the Planning Act.

As the Lakeshore Capacity Study was concerned with the impact of cottages on the environment, it was essential to include cottages used all year as well as those used intermittently. Thus, the term "cottage" was used in a broad sense to include all single-family residential buildings on the lakeshore. As lakeshore development in the Study Area was primarily single-tier development, most cottages had lake frontage. In the few situations where cottages were located in two or three tiers, all were included.

A geographic cluster sampling technique was chosen as the most efficient way of deriving regional estimates. One-ninth of each township (about 2080 hectares) was selected as the most appropriate cluster size and a grid was drawn on that basis. Where the townships were irregular in shape, the uniform grid was extended to provide clusters of similar size. This produced a total of 378 clusters, from which 40 were initially selected on a random basis as sample blocks.

The total number of cottages to be surveyed was determined in relation to the estimated number of cottages in the Study Area. Based on published reports and contacts with regional and county officials, it was estimated that there were about 35,000 cottages in Muskoka and Haliburton. From this figure, it was calculated that a minimum of 3,000 cottages should receive questionnaires in order to ensure a sufficient number of valid responses.

The next step was cottage enumeration. Starting with the sample block selected first, and proceeding in sequence, field staff located and numbered each cottage in each sample block. The required minimum of 3,000 cottages was reached in the 38th sample block selected. The remaining cottages in that sample block were then recorded, to produce a total enumeration of 3,280 cottages.

Questionnaires were distributed to the enumerated cottages in July and on the Civic Holiday long weekend early in August. Later, at the end of August, call-backs were made to all non-respondents, to determine the number of "non-contacts" (cottages where the questionnaire had not been picked up) and to encourage a higher rate of response. As there were relatively few non-contacts (about 1%), it was assumed that they created no significant bias in the responses.

The call-backs were successful in generating a renewed wave of returns. A response rate of 39% was achieved, despite three interruptions in mail service during the period from late July to October.

The questionnaire was designed to collect cottage-use data for a full year, from July 1977 to June 1978. To make completion of the questionnaire as simple as possible, a calendar was provided and the cottage owners were asked to record on the calendar the number of people at the cottage on each day it was in use. From this information, it was possible to calculate both cottage-days and user-days for each cottage and, when aggregated, for each lake.

6.2 AUTUMN SURVEY OF LAKESHORE RESIDENTS

It was apparent from the 1978 Survey of Lakeshore Residents that most of the cottage use during the autumn, winter and spring was associated with either permanent use or long weekends. Therefore, it was assumed that cottagers who used the cottage intermittently could recall their use of the cottage with an acceptable degree of accuracy for these periods of relatively low use. The question remained, however, whether they could recall their cottage use accurately for the previous summer,

when they used the cottage more frequently. Accordingly, the Autumn Survey of Lakeshore Residents was designed to obtain cottage-use data for the most recent four months, from July to October 1978. This information would then be compared with the cottage use recorded for the same months of the previous year.

The questionnaire was mailed in November to each respondent who had provided his/her name and address on the summer survey return, as an indication of willingness to participate in a follow-up survey to up-date some of the information. Over 80% of the respondents had included the information necessary for mailing, which probably reflected their deep interest in their recreational properties. Whatever the reason, the response rate for the Autumn Survey soared to 74% (775 questionnaires).

In the Autumn Survey, cottagers were asked to record their cottage use for the previous four months, so the maximum length of recall was five months. This was substantially less than the recall period in the earlier survey, in which cottagers had been asked to record their cottage use for the previous twelve months. Thus, it was possible that the information recorded might be quite different.

For the cottages covered in the Autumn Survey, the daily cottage-days for July 1 to October 31, 1977, were plotted as a bar graph. Similarly, the daily cottage-days for July 1 to October 31, 1978, were plotted on a transparent overlay. When the position of the overlay was adjusted slightly so the weekends coincided, a remarkable similarity was evident in the pattern and level of cottage use. Further, the correlation coefficient between the cottage-day data for 1978 and 1977 was 0.75. As some variation in use can be expected to occur from year to year, even for the same cottages, the strength of this correlation affirmed the ability of respondents to recall their cottage use for the previous summer. Therefore, the 1978 Survey of Lakeshore Residents was considered to be a valid record of cottage use over the full 12 months, from July 1977 to June 1978.

7. HIGHLIGHTS OF SURVEY FINDINGS

In general terms, the pattern of cottage use in Ontario was well known. It was widely recognized, for example, that summer was the period of greatest use, that winter use was increasing and that many cottages built originally for seasonal use were now being used all year. The 1978 Survey of Lakeshore Residents provided much more specific information for the Study Area.

7.1 COTTAGE USE LEVELS

A record of the intensity of cottage use was obtained from the calendar in the questionnaire. Of particular interest was the information on the number of days per year the cottage was used. It seemed logical to assume that the carrying capacity of the lakeshore would be greater for cottages used intermittently than for those used all year. How much greater would depend in part on the amount of use.

As the level of cottage use varied with the season, the average use for each month was computed. The seasonal pattern was quite striking (Figure 4). It did not break into quarters but rather into a long winter season and a short spring, summer and fall. The low-use winter months were November to April. During this period, average monthly use of the cottage was only five days whereas in the high-use summer months, July and August, the monthly

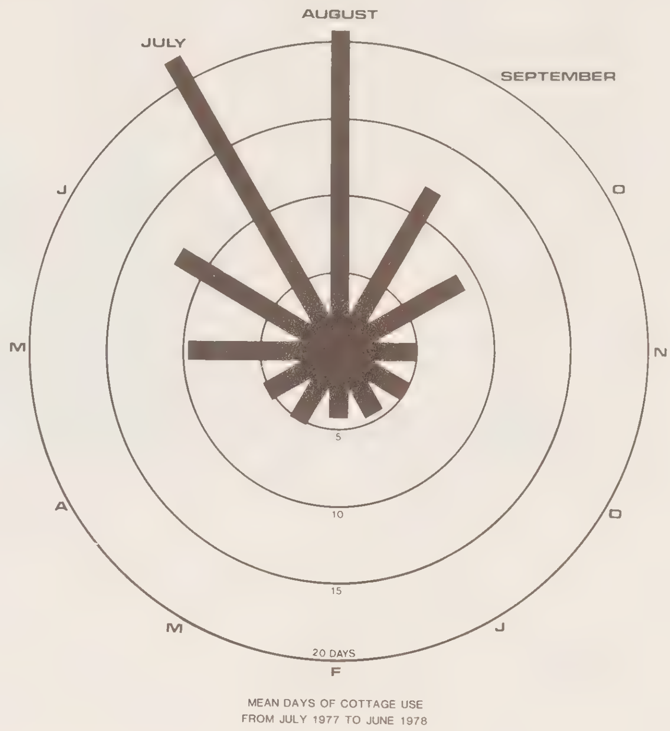


Figure 4. Cottage use, by month

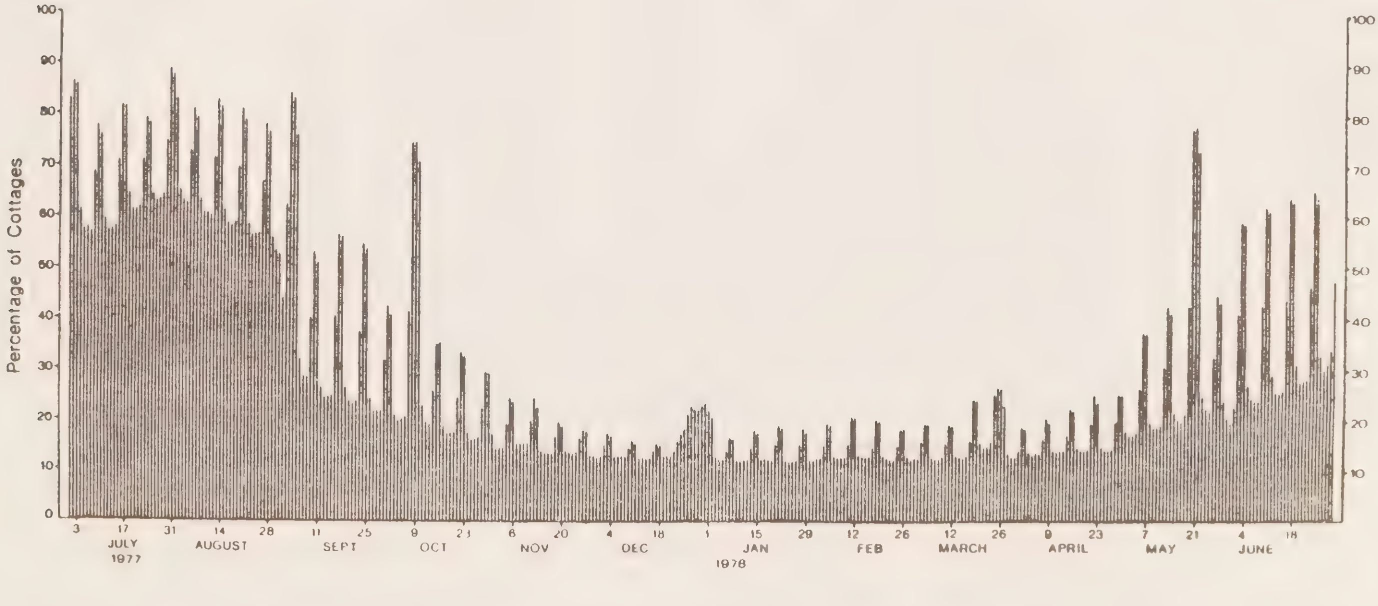
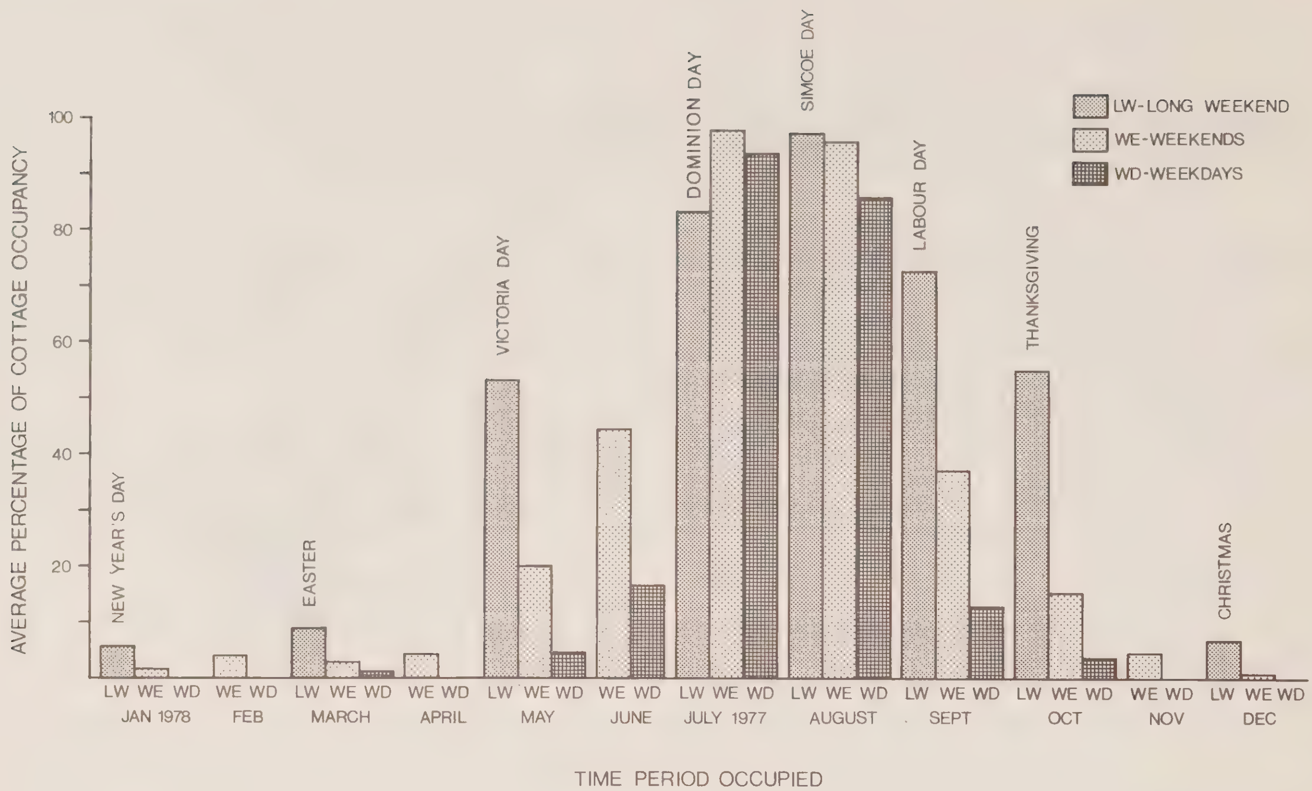


Figure 5. Percentage of cottages occupied in Muskoka-Haliburton, daily for the year ending June 30, 1978

SUMMER (JULY AND AUGUST)



WEEKENDS

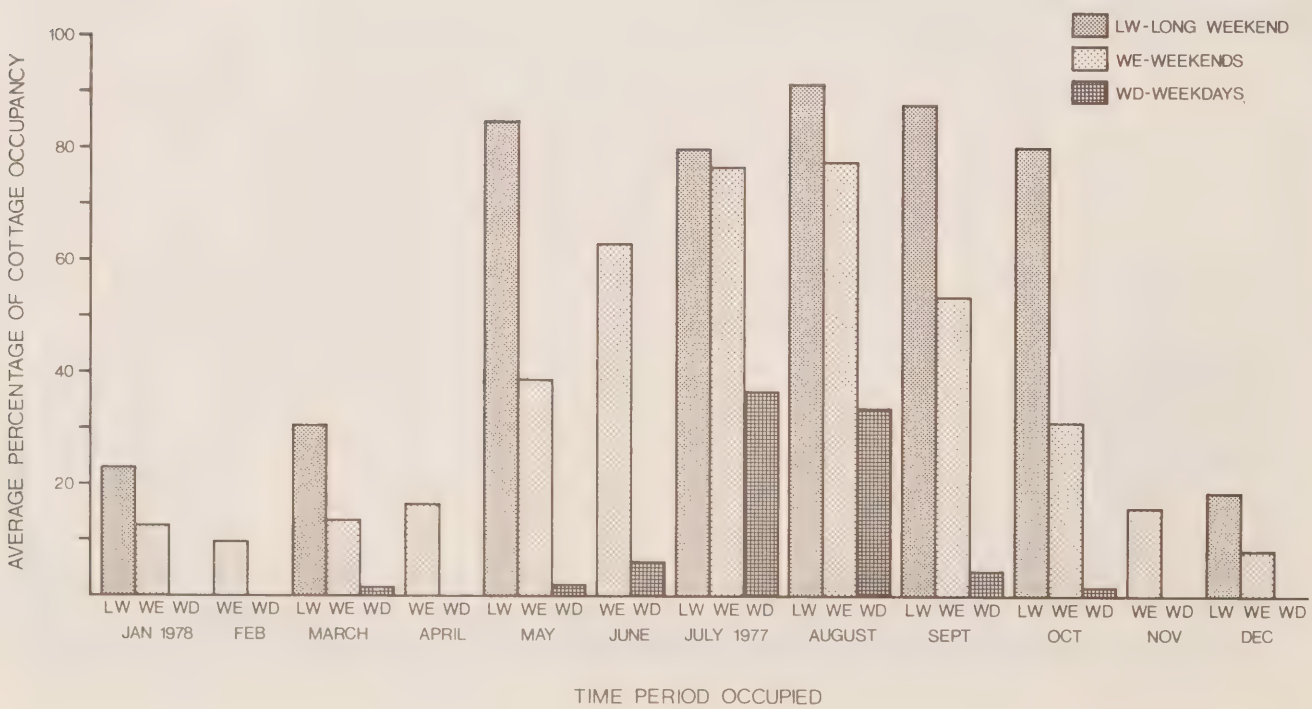
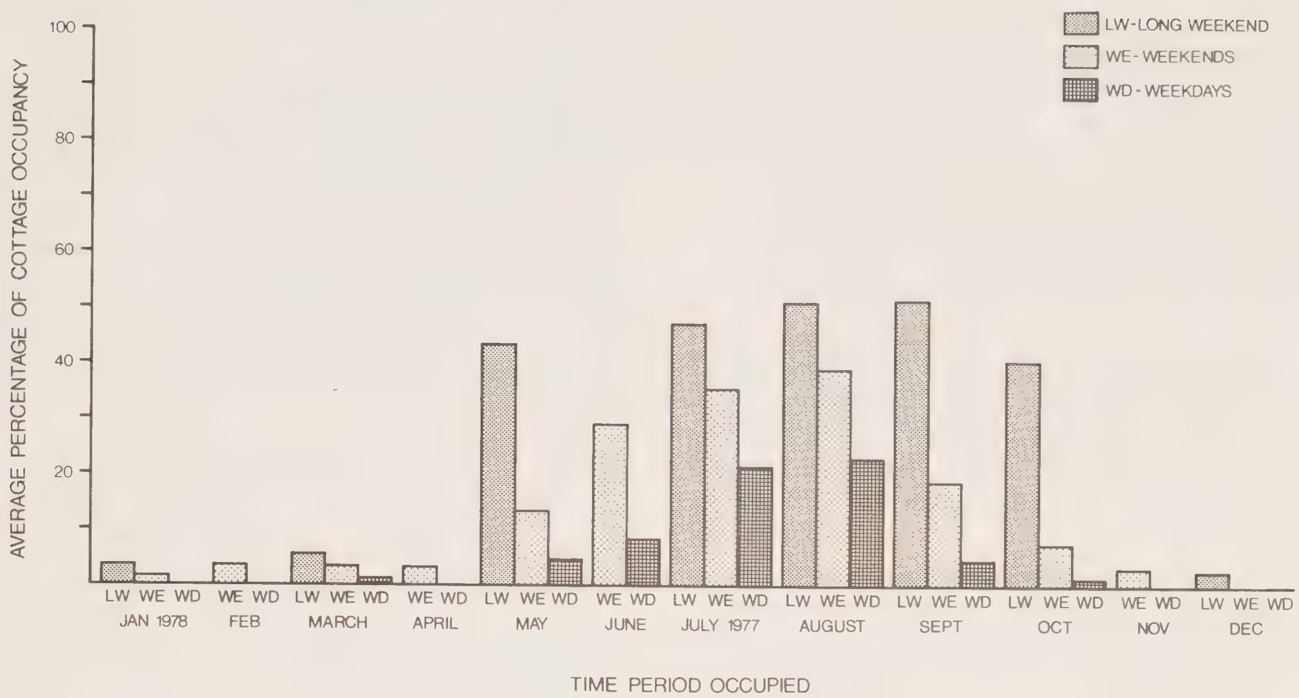


Figure 6. Cottage use patterns, by cottage typology group

LONG WEEKENDS



EXTENDED SUMMER

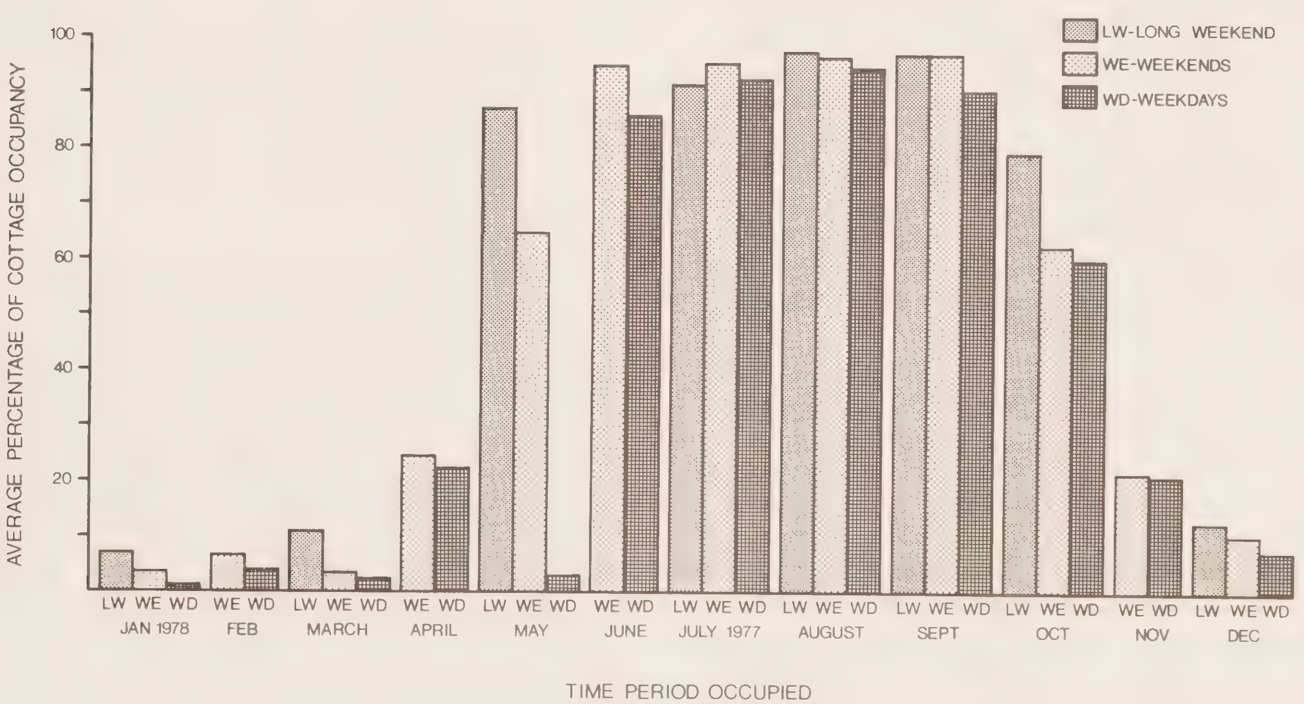


Figure 6. Cottage use patterns, by cottage typology group

average was twenty-two days. Spring and fall were transitional periods, with an average of twelve days in June and September and nine days in May and November. The cottages were in use for an average of 116 days during the year, which was roughly one-third of full-time use.

This seasonal pattern of cottage use was important to several study components. For example, the Fisheries Component research group had indicated that ice-fishing in the winter was hard on the trout fishery. Similarly, the Wildlife Component group had stated that loons were more vulnerable to human disturbance during their nesting period in June. So knowing the timing of the heavy influx of cottagers was relevant to the analysis and prediction of cottage impact on the environment.

The daily use data (Figure 5) confirmed that there was a consistent weekly pattern of cottage use, not only during the summer but also throughout the rest of the year. More cottages were used on weekends than on weekdays. Further, more cottages were used on long weekends and other holidays, such as Dominion Day, than on other weekends.

While this information provided an illuminating overview of cottage occupancy, it did not link the use patterns to individual cottages. To do this, a statistical analysis was undertaken to develop a typology of cottages.

7.2 COTTAGE USE PATTERNS

The purpose of the cottage typology was to classify the cottages according to their predominant use patterns, so the survey findings would provide a more realistic reflection of actual cottage use. It was known at this stage that the level of cottage use varied considerably among cottages, so it was not surprising that the global average of 116 cottage-days for the year seemed unrealistically high to some cottage owners. It was considered unlikely that each cottage was unique, so it was assumed that the cottages could be distinguished by their different patterns of predominant use and grouped accordingly. Cottage owners would then be able to identify a use pattern similar to their own and the average annual use for that particular group of cottages would be more likely to approximate their actual level of use.

In fact, the overall average was relatively high because the survey included cottages occupied all year and cottages occupied for part of the year as a principal residence. The latter group had been identified when the questionnaire for the 1978 Survey of Lakeshore Residents was pre-tested in the Study Area. Usually, the owners spent four to five months at the cottage and the rest of the year in a warmer climate. While these cottages comprised a small proportion of the total sample, they were important because each cottage had relatively high use.

The different types of cottages already observed included those used primarily on long weekends, those used all year, and those used for part of the year as a principal residence. In addition, it was known that some cottages were occupied most of the summer while others were used almost every weekend. It was expected that the statistical analysis would identify some of these types of predominant use and possibly others.

The cottage typology analysis is described in detail in the Land Use Component Technical Report on Cottage Typology (Miles, 1980). Some of the steps are noted briefly here, to assist in understanding the nature of the typology.

Initially, the days in use each month, for each cottage, were assigned to three separate categories: weekdays, weekends and long weekends. For example, when a cottage was occupied for the entire month of September, the thirty days were assigned as follows: four days to September long weekends, ten days to September weekends and sixteen days to September weekdays. For the entire year, this process required 32 variables, including 12 for weekdays, 12 for weekends and 8 for long weekends (as four months had no long weekends.)

Principal component analysis of the 32 variables describing the pattern of use for each cottage identified four distinct groups of cottages, each with similar use patterns. The next step was to compute four component scores for each cottage, to indicate the relationship of the cottage to each of the four groups. Cluster analysis of the component scores then distributed the cottages among five mutually exclusive groups.

Group 1. *Summer (July and August)*
Cottages used intensively during the summer (July and August) but much less at other times.

Group 2. *Weekends*
Cottages used predominantly on weekends in the spring, summer and fall.

Group 3. *Long Weekends*
Cottages with low levels of use, except on long weekends in the spring, summer and fall.

Group 4. *Extended Summer*
Cottages with high levels of use in the spring, summer and fall.

Group 5. *All Year*
Cottages used all year, as a principal residence.

The nature of each group is illustrated in a bar chart (Figure 6). When the charts are compared, the differences in the predominant patterns of use are readily apparent. The Summer (July and August) group had a high percentage of cottages occupied on both weekdays and weekends during July and August. In contrast, while the Extended Summer cottages were also occupied on weekdays and weekends, their use extended beyond the summer into the spring and fall. The Weekends and Long Weekends groups included cottages used in the spring, summer and fall; one group was characterized predominantly by weekend use and the other by long-weekend use. The occupancy rate for the All Year group was the same throughout the year, so a chart was unnecessary.

The cottage typology groups differed in their average cottage use and household size. Table 1 shows the percentage of the total sample of cottages in each

typology group and the cottage use (average number of cottage-days per year) and household size (average number of persons per cottage) for each group.

Nearly one-quarter of the cottages were used predominantly on long weekends from June to September. Their average use for the year was 39 days.

Cottages used predominantly during the summer (July and August) or on weekends during the spring, summer and fall, each comprised 28% of the sample. The Summer cottages were used for an average of 89 days per year and the Weekend cottages for an average of 81 days.

Cottages used all year, as a principal residence, were occupied for an average of 363 days, roughly four times as much as the Summer and Weekend cottages. Thus, the carrying capacity of the lake and lakeshore, in terms of number of cottages, is likely to be considerably greater for cottages used intermittently than for those used all year.

Table 1. Cottage use and household size for five cottage typology groups

COTTAGE TYPOLOGY GROUP	Percent of Total Sample	COTTAGE USE Average Cottage-days per Year	HOUSEHOLD SIZE Average Persons per Cottage
	%	no.	no.
Summer (July and August)	27.7	88.7	3.69
Weekends	28.0	80.5	3.60
Long Weekends	23.9	39.3	3.39
Extended Summer	7.8	175.2	2.64
All Year	12.6	363.1	2.56
All Cottages	100.0 (N=1139)	116.03	3.07

7.3 SUMMARY OF SURVEY FINDINGS

Several conclusions were drawn from the analysis of cottage use in the Muskoka-Haliburton area:

1. There is a consistent seasonal pattern of cottage use. Based on the monthly average for all cottages, the period of highest use extends for only two months, July and August; the period of lowest use extends for six months, from November to April. Transitional months occur both in the spring (May and June) and in the fall (September and October).

2. There is also a consistent weekly pattern of cottage use throughout the year. More cottages are occupied on weekends than on weekdays. Further, more cottages are occupied on long weekends and other holidays (such as Dominion Day) than on other weekends.

3. Cottages can be classified into five groups which are internally homogeneous and mutually exclusive with respect to their predominant use patterns.

   Group 1. Summer (July and August)
   Group 2. Weekends (spring, summer and fall)
   Group 3. Long Weekends (spring, summer and fall)
   Group 4. Extended Summer (weekends and weekdays)
   Group 5. All Year

4. The cottage groups differ in level of cottage use. The average annual number of cottage-days for each group is:

Group 1. Summer	88.7
Group 2. Weekends	80.5
Group 3. Long Weekends	39.3
Group 4. Extended Summer	175.2
Group 5. All Year	363.1
All Cottages	116.03

   Thus, cottages used all year, as a principal residence, are occupied about four times longer (363 days) than cottages in the Summer (89 days) or Weekends (81 days) groups.

5. The cottage groups differ also in household size. The average number of persons per cottage is:

Group 1. Summer	3.69
Group 2. Weekends	3.60
Group 3. Long Weekends	3.39
Group 4. Extended Summer	2.64
Group 5. All Year	2.56
All Cottages	3.07

   Thus, cottages in the Summer, Weekends and Long Weekends groups have more occupants than the average, while cottages in the Extended Summer and All Year groups have fewer occupants than the average.

6. The differences in cottage-days and persons per cottage are reflected in the average annual user-days for each group.

Group 1. Summer	327.3
Group 2. Weekends	289.8
Group 3. Long Weekends	133.2
Group 4. Extended Summer	462.5
Group 5. All Year	929.5
All Cottages	356.2

   Despite their smaller household size, cottages in the All Year group have by far the largest average number of user-days during the year. The figure for this group (930 user-days) is roughly double that of the Extended Summer group (463 user-days) and more than triple that of the Weekends group (290 user-days).

8. CONCEPTUAL APPROACH TO THE COTTAGER MODEL

The pattern of cottage use recorded for each cottage in the 1978 Survey of Lakeshore Residents represented the end result of all the decisions made by the cottagers regarding the use of their cottage. When viewed in this way, it was evident that the separate groups in the cottage typology provided a valid basis for predicting cottage use, as they reflected not only decisions influenced by the socio-economic characteristics of cottage owners but also decisions in response to changes in external factors, such as fuel costs and property taxes.

The average annual levels of cottage use for the five typology groups extended over a wide range, from 39 cottage-days for cottages in the Long Weekends group to 363 cottage-days for cottages in the All Year group. These differences offered a compelling rationale for employing the cottage groups as predictors.

Such an approach required information on the proportion of cottages in each typology group in the proposed development. As this would not be known at the time of the prediction, a method of estimating the proportions was needed. Accordingly, it was decided to introduce a system of benchmarking, whereby the cottage typology proportions for the "target" lake (where development was proposed) would be derived from a "benchmark" lake.

The logic for employing benchmark lakes was similar to the rationale for "hot-decking" in survey research. The assumption was that cases with similar values for measured variables (e.g., accessibility characteristics) would also have similar values for unmeasured variables (i.e., cottage typology). The feasibility of the benchmarking rested on the capacity of these measured variables to provide a close fit between the ultimate cottage typology on the target lake and the known typology on its benchmark lake.

The justification for employing the benchmarking process was that the cottage typology data could not be applied without it. Benchmarking provided the missing link (i.e., cottage typology proportions) that enabled the new knowledge of cottage use gained from the typology analysis to be applied in a practical method of predicting cottage use.

By definition, the benchmark lakes were those for which the cottage typology was known, that is, those included in the 1978 Survey of Lakeshore Residents. From this pool of possible benchmarks, the most appropriate lake would be selected. The proportion of cottages in each typology group, for the target lake, would then be assumed to be the same as for its benchmark lake.

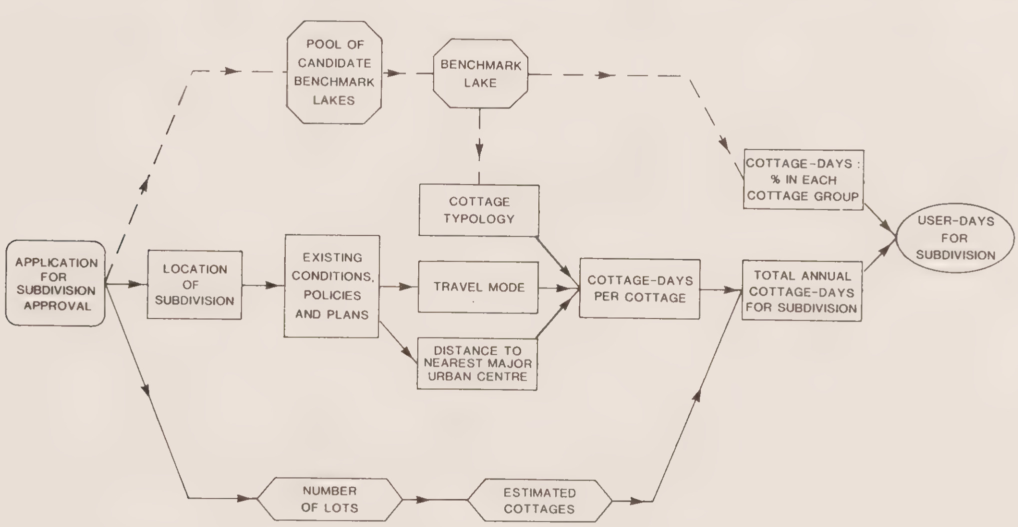


Figure 7. Cottager Model: a conceptual diagram linking cottage typology and cottage use

To assist in selecting an appropriate benchmark lake, the official plan policies and zoning regulations with respect to seasonal versus permanent use of cottages would be examined. For example, if permanent use was not permitted on the target lake, the benchmark lake would have to be one with no cottages in the All Year group. In contrast, if the municipality had no planning policies regarding cottage use, it would be assumed that some cottages would be used all year and this would have to be reflected in the typology of the benchmark lake.

The conceptual framework for the Cottager Model is shown in the network diagram in Figure 7. While the model resembled the Accessibility Model in its overall structure, it differed in employing the cottage typology groups as the major predictors.

The other major difference from the Accessibility Model was the benchmarking process. The proportion of cottages in each typology group, for the target lake, would be derived from its benchmark lake. Subsequently, at the stage when the predicted cottage-days were to be converted to user-days, the proportion of cottage-days in each typology group, for the target lake, would be derived from its benchmark lake. By allocating the total cottage-days for the subdivision to the typology groups, it would be possible to convert to user-days by applying the appropriate household size for each group.

In summary, the Cottager Model would take into account variations in cottage use between lakes, by recognizing differences in accessibility characteristics and planning policies when selecting the benchmark lake. The model would also reflect differences in cottage use between cottages on the same lake, by employing the cottage typology groups as variables. In addition, variations in household size would be taken into account by converting cottage-days to user-days for each typology group.

Overall, the Cottager Model concept incorporated several refinements which were not in the Accessibility Model. While most of these were designed to improve the predictions, the model had to rely on the benchmarking process for estimates of cottage typology. Thus, some of the gains in accuracy would be offset by losses inherent in the benchmarking.

9. ACCESSIBILITY MODEL

The Accessibility Model and the Cottager Model were both designed to predict cottage use. The Accessibility Model derives its predictive power from variations in cottage accessibility, while the Cottager Model exploits cottage use patterns.

The conceptual approach to the Accessibility Model was described in Chapter 4. The development of the model is outlined in this chapter, which includes selected supporting data. More technical detail regarding the modelling process appears in the Technical Report on Land Use Models to Predict Cottage Use (Carroll et al., 1982).

9.1 CANDIDATE VARIABLES

In the conceptual approach to the Accessibility Model, four aspects of cottage access were identified as possible predictors of cottage use. Their selection was based on the literature search and on observation of the Study Area. These four aspects were: type of access to the cottage, access road open in winter, distance travelled on poor roads to reach the cottage, and distance from the cottage to the nearest urban centre. From these general descriptions, nine candidate variables were defined and computed (Table 2).

The type of access to each cottage was expressed in terms of four travel mode variables. Travel Mode 1 was defined as car in summer, car in winter; Travel Mode 2 as car in summer, other in winter; Travel Mode 3 as boat in summer, other in winter; and Travel Mode 4 as no winter use. The percentage of cottages with each travel mode is shown in Table 3.

The travel modes were dichotomous variables so they were treated as dummy variables. For example, for a cottage with water access only, the travel mode variable specifying "boat in summer, other in winter" would be assigned a value of 1 and the other travel modes would each have a value of 0.

The variable "access road open in winter" was also a dichotomous variable, treated as a dummy variable.

To measure the distance travelled on poor roads to reach the cottage, a "poor" road was defined as one that was not hard-surfaced. While this might appear at first glance to be too simplistic, distinguishing between hard-surfaced and other roads reflected indirectly such specific characteristics as the number of travel lanes, radius of curvature and gradient, each of which would affect travel time. As this was a continuous variable, its value was the measured distance (in kilometres) from a hard-surfaced road to the cottage.

Table 2. Candidate variables for the Accessibility Equation

CONTINUOUS VARIABLES	
DISTSUMH	Distance (in km) from the cottage to the nearest high order service centre, in summer.*
DISTSUML	Distance (in km) from the cottage to the nearest low order service centre, in summer.*
INTERACT	Interaction (cross-product) between DISTSUMH and DISTSUML.
MAXPOOR	Distance (in km) from the cottage to the nearest good (hard-surfaced) road, i.e. distance on poor roads.**
DICHOTOMOUS VARIABLES	
TMODE1	Mode of travel to the cottage: car in summer; car in winter.
TMODE2	Car in summer; other in winter (e.g. snowmobile, skiis, foot).
TMODE3	Other in summer (e.g. boat); other in winter.
TMODE4	Car or other in summer; no winter use.
NOPLOW	Access road to the cottage closed in winter.

*Variables DISTWINH and DISTWINL (distance in winter) were removed from the candidate variables, due to their high correlations with DISTSUMH and DISTSUML, respectively.

**Bivariate relationships among the independent variables and between each of the independent variables and the dependent variable (cottage-days) were explored, to determine whether any of the assumptions of regression modelling had been violated. The only adjustment necessary was transformation of the variable 'distance on poor roads' to its reciprocal (MAXPOORX) to produce a more linear relationship with cottage-days. To avoid dividing by zero when the distance on poor roads was zero, 0.003 was added to MAXPOOR. Thus, the new variable MAXPOORX is equivalent to $\frac{1}{\text{MAXPOOR} + 0.003}$

Table 3. Travel mode variables

Computer Mnemonic	Travel Mode	Number of Cottages	Percent of Cottages
TMODE1	Car in Summer Car in Winter	394	34.59
TMODE2	Car in Summer Other in Winter	231	20.28
TMODE3	Boat in Summer Other in Winter	88	7.73
TMODE4	Car or boat in Summer No Winter Use	397	34.85

To measure the distance from the cottage to the nearest urban centre in the region, it was essential to identify botl high order and low order urban service centres in the Study Area. The method of selection was based on Christaller's theory of central place, where the central place provides goods and services for people in the surrounding area (Christaller, 1933). The distinction between "high order" and "low order" relates to the number and diversity of goods and services provided, the size of the population and the extent of the tributary area. High order centres could serve a dual role for people in the immediate vicinity, as their wider variety of goods and services includes low order goods and services.

The hierarchy of central places ranges from large metropolitan complexes to small towns and villages. However, in the Study Area the urban centres were all at the lower end of the scale. The largest town was Huntsville, which had a population of 10,480 in 1976. Thus, the high order centres selected were mainly towns and the low order centres were mainly villages.

The initial selection of urban centres was based on population, the number of retail establishments and the diversity of goods and services. This was compared with responses to a question in the survey regarding the urban centres visited most frequently by members of the cottage household. The responses from the cottagers confirmed the initial selection of both high and low order centres.

Table 4. High and low order urban service centres in the Muskoka-Haliburton Study Area

HIGH ORDER URBAN SERVICE CENTRES

Urban Centre	County or District Municipality	Population* 1971	Population* 1976
Bracebridge	Muskoka	6,903	8,180
Gravenhurst	Muskoka	7,133	7,350
Huntsville	Muskoka	9,784	10,840
Haliburton	Haliburton	899	1,124
Minden	Haliburton	697	590
Bancroft**	Hastings	2,276	2,210

LOW ORDER URBAN SERVICE CENTRES

Urban Centre	County or District Municipality	Population* 1971	Population* 1976
Bala	Muskoka	462	536
Baysville	Muskoka	283	240
Coldwater	Muskoka	759	803
Dorset	***	n/a	n/a
Mactier	Muskoka	794	690
Port Carling	Muskoka	617	628
Washago	Muskoka	423	442
Wilberforce	Haliburton	229	266

*Source: 1976 Statistics Canada, Population Statistics
**Outside Study Area but serves cottages in the southeast corner of Study Area
***Located on the boundary between Muskoka and Haliburton

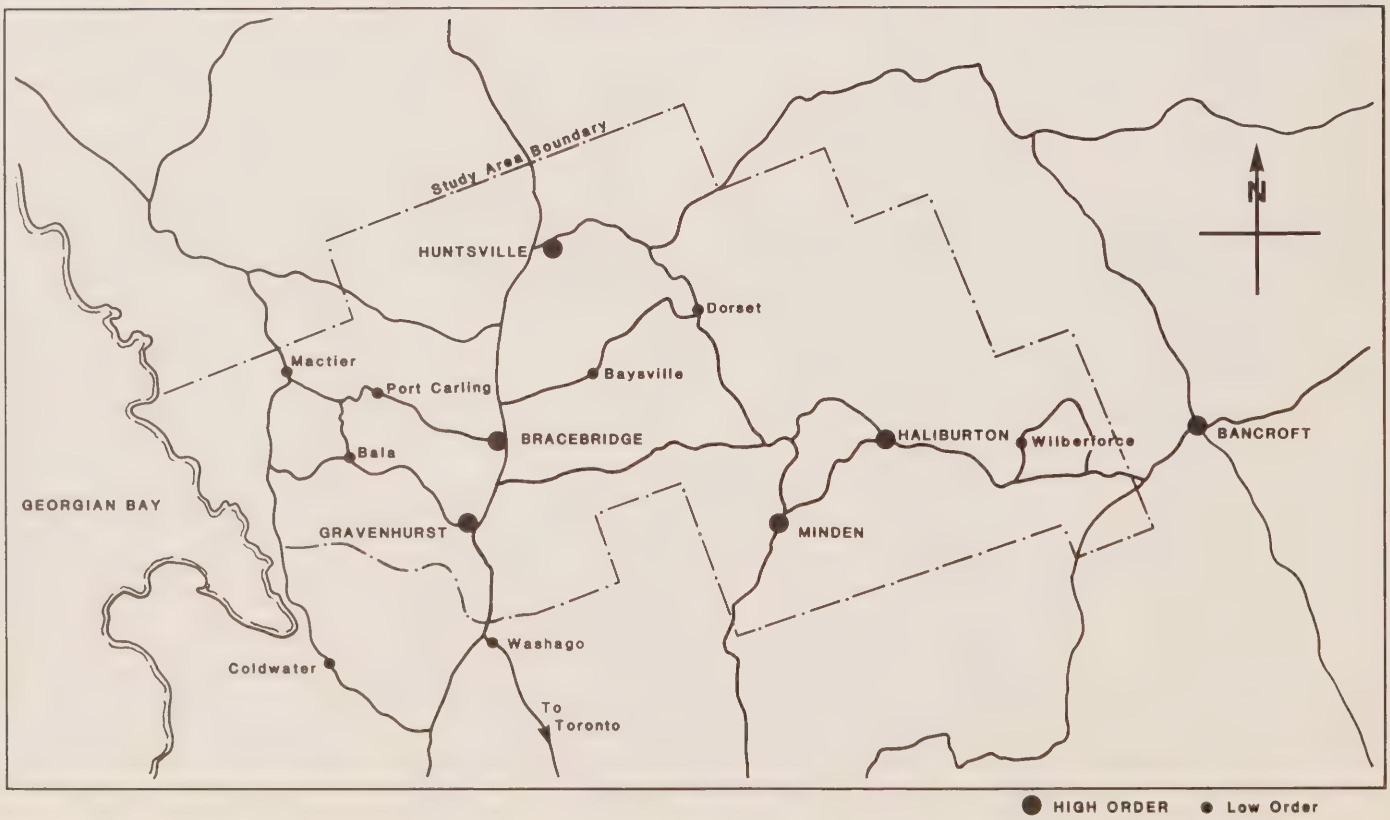


Figure 8. Urban centres in Study Area

They also indicated that cottagers in the southeast corner of the Study Area used Bancroft, which was outside the Study Area, as a high order service centre.

The final selection of urban centres appears in Table 4, together with the most recent population figures at the time the selection was made. These urban centres were used to measure the distance in kilometres from a cottage to its nearest high order urban centre and to its nearest low order urban centre (Figure 8).

The hypothesis that the independent variables were not causally related to the dependent variable (cottage-days) was tested by examining the Pearson product-moment coefficient of correlation between the dependent variable and each of the independent variables. In each case, the null hypothesis was rejected (Table 5). While this did not prove that a true relationship existed between cottage-days and the independent variables selected, it did indicate that such a relationship was possible.

Examination of the correlation matrix indicated that two of the independent variables had a moderately strong relationship with the dependent variable (annual cottage-days). The correlation between cottage-days and the travel mode variable signifying road access in summer and winter, was 0.501; while the correlation between cottage-days and the reciprocal of the distance on poor roads was 0.418. Each of the other independent variables showed a weak but statistically significant correlation with cottage-days.

9.2 REGRESSION EQUATION

The regression equation for the Accessibility Model was developed through multivariate analysis, employing stepwise regression procedures to identify the best equation. No problems were encountered with technique-dependent results, as the five procedures produced the same equations (Table 6).

The proportion of total variance in annual cottage-days explained by the regression equation and measured by the coefficient of multiple determination (R^2) increased with the addition of another variable in each of the first five steps but showed trivial gains in explained variance thereafter. Similarly, the mean square error (MSE) decreased in each step up to the five-variable equation but remained almost constant thereafter. In addition, the C_p

Table 6. Accessibility Model regression equations identified by five stepwise procedures*

Number of Variables in Equation	Variable Added in this Step**	Coefficient of Multiple Determination	C_p	Mean Square Error
1***	TMODE1	0.250	150.57	7912.5
2***	MAXPOORX	0.328	19.39	7099.0
3***	DISTSUMH	0.332	14.05	7060.0
4***	TMODE2	0.337	8.19	7017.7
5***	TMODE3	0.340	5.20	6993.1
6	DISTSUML	0.341	5.01	6985.8
7	INTERACT	0.341	7.00	6991.9
8	NOPLOW	0.341	9.00	6998.1

*Forward, Backward, Stepwise, Maximum R^2, Minimum R^2
**Or deleted from next step, as in Backward Elimination procedure.
***All variables statistically significant at or above 0.05 level.

closely approximated the number of variables in the equation at the fifth step, thereby identifying that equation as the one with the least number of variables which could predict almost as accurately as the equation using all possible predictors. All of these reasons contributed to the choice of the following equation as the best solution.

$$R_A = 83.105 + 93.452\,\text{TMODE1} + 0.322\,\text{MAXPOORX} - 5.29\,\text{DISTSUMH} + 22.655\,\text{TMODE2} + 21.972\,\text{TMODE3}$$

where R_A = number of annual cottage-days per cottage predicted by the Accessibility Model regression equation

TMODE1 = 1 when the travel mode is "car in summer, car in winter" (otherwise = 0)

MAXPOORX= the reciprocal of the distance in kilometres from the cottage to the nearest hard-surfaced road

DISTSUMH = the distance in kilometres from the cottage to the nearest high order urban centre

TMODE2 = 1 when the travel mode is "car in summer, other in winter" (otherwise = 0)

TMODE3 = 1 when the travel mode is "boat in summer, other in winter" (otherwise = 0)

The two variables "car in summer, car in winter" and "the reciprocal of the distance from the cottage to the nearest hard-surfaced road" together explained nearly one-third of the variance in annual cottage-days (R^2 = 0.328). This was not surprising in view of the correlation between each of these variables (TMODE1 and MAXPOORX) and cottage-days, and their relative lack of intercorrelation (Table 5).

As expected, the regression coefficient for "distance to the nearest high order urban centre" was negative, confirming that a greater distance would predict a lower level of cottage use when the other variables were held constant (Table 7).

When the Student's t test was applied as each independent variable was entered into the equation, the hypothesis that no relationship existed between the annual number of cottage-days and the independent variable was rejected.

The high value of the F ratio ($F = 116.77$) revealed that the explained portion of the variance in the annual number of cottage-days was substantially greater than the unexplained portion. Further, the high significance level ($p < .0001$) meant that the regression was unlikely to be a product of chance.

When the values of the predicted error sum of squares

Table 5. Correlation matrix for candidate variables in the Accessibility Equation

	ANNUAL COTTAGE-DAYS	DISTSUMH	DISTSUML	INTERACT	MAXPOOR	MAXPOORX	NOPLOW	TMODE1	TMODE2	TMODE3	TMODE4
ANNUAL COTTAGE-DAYS	1	0.250	−0.194	−0.237	−0.262	0.418	0.213	0.501	−0.139	−0.083*	−0.338
DISTSUMH		1	0.595	0.843	0.663	−0.222	−0.209	−0.299	0.213	0.122	0.057**
DISTSUML			1	0.897	0.378	−0.183	−0.201	−0.329	0.142	0.168	0.113
INTERACT				1	0.553	−0.207	−0.214	−0.348	0.181	0.176	0.096
MAXPOOR					1	−0.553	−0.246	−0.306	0.206	0.122	0.062*
MAXPOORX						1	0.381	0.306	−0.154	−0.089	−0.148
NOPLOW							1	0.249	−0.337	0.224	−0.097
TMODE1								1	−0.365	−0.210	−0.531
TMODE2									1	−0.145	−0.368
TMODE3										1	−0.212
TMODE4											1

N = 1139
Listwise deletion employed
All correlations significant at the 0.001 level or above, except those with an asterisk.
Significance of those denoted * lies between 0.05 and 0.001
Correlations denoted ** are not significant.

(PRESS) and the error sum of squares (SSE) were compared, it was evident that they were similar, indicating that the regression equation could predict the annual number of cottage-days almost as well for a different sample of cottages drawn from the same area.

The final step in evaluating the regression equation was to examine the pattern of residuals in relation to the regression line. As this is described and illustrated in the Technical Report on Land Use Models, it is sufficient to note here that there were no extreme outliers, no marked tendency for the distribution of residuals to depart from normality and no evidence of heteroscedasticity. In short, the assumptions of regression analysis were satisfied.

Together, the five variables in the regression equation explained 34% ($R^2 = 0.340$) of the variance in annual cottage-days among the 1,139 cottages for which responses were received.

Table 7. Regression equation selected for Accessibility Model

Annual Cottage-days = 83.105 + 93.452 TMODE1 + 0.322 MAXPOORX − 0.529 DISTSUMH + 22.655 TMODE2 + 21.972 TMODE3

Variable	Regression Coefficient	Standard Error	Student's t	p
TMODE1	93.452	6.127	15.2	.001
MAXPOORX	0.322	0.029	11.2	.001
DISTSUMH	−0.529	0.166	3.2	.01
TMODE2	22.655	6.907	3.3	.01
TMODE3	21.972	9.837	2.2	.05

Coefficient of Multiple Determination (R^2)	= 0.340
Analysis-of-variance F-ratio (F)	= 116.77 ($p < .0001$)
Error Sum of Squares (SSE)	=7923279
Predicted Error Sum of Squares (PRESS)	=8013450
Mallory's C_p (C_p)	= 5.208
Mean Square Error (MSE)	=6993.1

9.3 PREDICTED COTTAGE-DAYS FOR SUBDIVISION

The regression equation for the Accessibility Model was designed to predict the annual number of cottage-days for one cottage. Where all the cottages in a subdivision have similar accessibility characteristics, the prediction will be the same for each cottage in the subdivision.

As zoning by-laws usually permit only one principal building on a registered lot, it was assumed that each proposed lot represented one potential cottage. Therefore, the number of cottages in a proposed subdivision would be the same as the number of lots.

To apply the regression equation to a given subdivision, the procedure is to multiply the predicted annual number of cottage-days per cottage by the number of lots in the subdivision. For example, if there are 55 lots in the subdivision and if 80 cottage-days per cottage are predicted by the regression equation, the total annual number of cottage-days can be calculated by multiplying 80 x 55 which equals 4,400 cottage-days.

In subdivisions where the type of access differs for some of the lots, the above procedure must be modified to reflect the different levels of cottage use associated with different types of access. If, for example, some lots are on the lakeshore and have road access all year while others are on an island and have water access only, the first step is to separate the lots into two categories according to their type of access. The procedure described above is applied to each category. The totals for the two categories can then be added together, to obtain the predicted cottage-days for the subdivision. In a similar manner, subdivision totals can be summed to yield predictions for the entire lakeshore.

Thus, the same procedure can be followed, whether the prediction is for a portion of a subdivision, an entire subdivision or all subdivisions on a lake, but if there are differences in the accessibility characteristics of lots it will be necessary to disaggregate before predicting and re-aggregate after predicting.

9.4 CONVERSION TO USER-DAYS

The Ontario Lakeshore Capacity Simulation Model employs user-days as one of the predictors of the amount of anthropogenic phosphorus likely to be contributed to the lake from cottages on the lakeshore. It was necessary, therefore, to convert the predicted cottage-days to user-days.

Data from the 1978 Survey of Lakeshore Residents indicated that cottages were used for an average of 116.03 cottage-days and 355.86 user-days annually. When the average number of user-days was divided by the average number of cottage-days, the average number of persons per cottage was found to be 3.07.

To convert cottage-days to user-days, the procedure is to multiply the predicted number of cottage-days by the average number of persons per cottage (3.07). If, for example, 4,400 cottage-days are predicted for the subdivision, the conversion to user-days would be calculated by multiplying 4,400 x 3.07 which equals 13,508 predicted annual user-days for the subdivision.

9.5 THE MODEL

The network diagram depicting the conceptual approach to the Accessibility Model (Figure 3) provided an overview of the model described in this chapter.

The Accessibility Model can also be expressed in the form of an equation:

PREDICTED ANNUAL USER-DAYS FOR SUBDIVISION = $R_A * N * T_x$

where $*$ = multiplication symbol

R_A = number of annual cottage-days per cottage predicted by the Accessibility Model regression equation

N = number of lots in subdivision

T_x = average number of persons per cottage (3.07) for all cottages in the LUC Survey

10. COTTAGER MODEL

The Cottager Model provides a second method of predicting cottage use. Its capability is based largely on patterns of cottage use, expressed as the proportion of cottages in each typology group. As the data base differs from that employed in the Accessibility Model, the two models can be considered complementary.

Initially, the regression equation for the Cottager Model was developed using the five cottage typology groups, that is, Summer, Weekends, Long Weekends, Extended Summer and All Year. The resulting equation explained almost 95% ($R^2 = 0.949$) of the variation in annual cottage use, for cottages in the survey sample for which the typology was known. However, as several calculations were required for each typology group, the model was complicated. For this reason, the possibility of combining some of the groups was explored, as a means of producing a model that would retain its predictive power but be easier to apply.

Cottages in the All Year group had the highest annual use – an average of 363 cottage-days. Thus, it was evident that an error in estimating the proportion of cottages in this group could have a considerable effect on the prediction. In contrast, redistribution of cottages among the groups with low annual use would have relatively little effect on the prediction. Therefore, the Summer (89 cottage-days), Weekends (81 cottage-days) and Long Weekends (39 cottage-days) groups were merged into a new group, which was termed "Seasonal". Cottages in this larger group had an average of 71 cottage-days annually.

A simplified Cottager Model was then developed, based on the three typology groups: Seasonal, Extended Summer and All Year. The regression equation explained 92% ($R^2 = 0.920$) of the variation in cottage use among cottages in the survey sample. While this represented a reduction in explanatory capability, the difference was considered negligible when compared to the larger losses inherent in the benchmarking process. Accordingly, the simplified version was chosen as the more practical Cottager Model.

10.1 CANDIDATE VARIABLES

In the conceptual approach to the Cottager Model (Chapter 8), the groups of cottages identified in the cottage typology (Chapter 7) were selected as possible predictors of cottage use. Underlying this choice was the assumption that the pattern of cottage use reflected the many decisions made by cottagers regarding the use of their cottages. Further, it was assumed that the decisions of other cottagers on the same lake, both now and in the future, would result in similar patterns of use.

The three cottage typology independent variables (Seasonal, Extended Summer and All Year) are defined in Table 8.

Examination of Table 9 indicates the wide variation among the groups in average annual cottage-days, the dependent variable in the Land Use models. The All Year group averaged 363 cottage-days, or about five times more than the Seasonal group (71 cottage-days). Between these two extremes, the Extended Summer group recorded an average of 175 cottage-days, equivalent to about half the year. These differences in annual use gave the cottage typology groups their potential as predictors.

The line graphs in Figure 9 record the percentage of annual user-days in each month, for each typology group. As expected, three distinct patterns emerged. Cottages in the Seasonal group had their peak use in July and August, with over 50% of their annual user-days concentrated in these two months. In contrast, cottages in the Extended Summer group were used over a longer period, with about 40% of their annual user-days in July and August, about 30% in June and September and about 20% in May and October. Cottage use for the All Year group was spread more evenly over the twelve months.

Table 8. Candidate variables for the Cottager Equation

Variable	Variable Description	Computer Mnemonic
COTTAGE TYPOLOGY VARIABLES		
Seasonal	Cottages used intensively during the summer (July and August); predominantly on weekends in the spring, summer and fall; and predominantly on long weekends in the spring, summer and fall.	SEASONAL
Extended Summer	Cottages with high levels of use throughout the spring, summer and fall.	ESUMMER
All Year	Cottages used all year, as a principal residence.	PERM
ACCESSIBILITY VARIABLES		
All variables listed in Table 2		

Table 10. Correlation matrix for candidate variables in the Cottager Equation

	ANNUAL COTTAGE-DAYS	DISTSUMH	DISTSUML	INTERACT	MAXPOOR	MAXPOORX	NOPLOW	TMODE1	TMODE2	TMODE3	TMODE4	SEASONAL	ESUMMER	PERM
ANNUAL COTTAGE-DAYS	1	0.250	−0.194	−0.237	−0.262	0.418	0.213	0.501	−0.139	−0.083*	−0.338	−0.866	0.167	0.915
DISTSUMH		1	0.595	0.843	0.663	−0.222	−0.209	−0.299	0.213	0.122	0.057**	0.196	−0.029**	−0.214
DISTSUML			1	0.897	0.378	−0.183	−0.201	−0.329	0.142	0.168	0.113	0.132	0.027**	−0.182
INTERACT				1	0.553	−0.207	−0.214	−0.348	0.181	0.176	0.096	0.173	−0.004**	−0.206
MAXPOOR					1	−0.553	−0.246	−0.306	0.206	0.122	0.062*	0.216	−0.052**	−0.221
MAXPOORX						1	0.381	0.306	−0.154	−0.089	−0.148	−0.350	−0.027**	0.446
NOPLOW							1	0.249	−0.337	0.224	−0.097	−0.179	−0.017**	0.232
TMODE1								1	−0.365	−0.210	−0.531	−0.374	−0.005**	0.457
TMODE2									1	−0.145	−0.368	0.130	0.008**	−0.165
TMODE3										1	−0.212	0.041**	0.051*	−0.091
TMODE4											1	0.243	−0.021**	−0.278
SEASONAL												1	−0.574	−0.750
ESUMMER													1	−0.111
PERM														1

N = 1139
Listwise deletion employed
All correlations significant at the 0.001 level or above, except those with an asterisk.
Significance of those denoted * lies between 0.05 and 0.001
Correlations denoted ** are not significant.

Also included as candidate variables for the Cottager Model were the nine accessibility variables defined in Table 2. This was done in order to explore the predictive capability of a composite model.

The three cottage typology variables were treated as dummy variables. The number of cottages in each typology group was expressed as a proportion of all the cottages in a proposed subdivision. Thus, the total value for all typology groups was 1. For example, if 20 cottages out of a total of 80 were in the All Year group, the value for that group would be 0.25. But if there were no permanent residents, the All Year group would be assigned a value of 0.

The correlation coefficients between the dependent variable (average annual cottage-days) and each independent variable in the Cottager Model appear in Table 10. As the accessibility variables were considered in Chapter 9, only those correlations involving the three cottage typology variables need to be examined here. A significant negative correlation (-0.866) was evident between cottage-days and the Seasonal group, which comprised cottages with relatively low annual use. In contrast, the cottage typology variables identifying cottages occupied either extensively throughout the spring, summer and fall (Extended Summer) or by permanent residents (All Year) manifested positive correlations with cottage-days (0.167 and 0.915, respectively). These results pointed up, in the language of correlation analysis, the substantial differences in average annual cottage-days among the cottage groups.

The reason for the strong relationship (0.915) between the variable identifying the All Year group and annual cottage-days was the bimodal character of the distribution of cottage-days (Figure 10). The two modes – one at 65 cottage-days and the other at 365 cottage-days – were associated with the Seasonal and All Year groups, respectively. The large difference between these central tendencies was captured in distinguishing the All Year cottages as a group. Clearly, the cottage typology groups, particularly All Year, were powerful predictors of cottage-days.

Another correlation of interest was that between the All Year typology variable and the accessibility variable indicating year-round car access to the cottage. The coefficient of 0.457 revealed a tendency for permanent residents to occupy cottages with accessibility characteristics favorable to year-round use. It was relationships of this kind between objective accessibility features and patterns of cottage use that made it feasible in principle to estimate the cottage typology proportions by employing a benchmarking process.

Table 9. Cottage use and household size for three cottage typology groups

COTTAGE TYPOLOGY GROUP	Percent of Total Sample	COTTAGE USE Average Cottage-days per Year	HOUSEHOLD SIZE Average Persons per Cottage
	%	no.	no.
Seasonal	79.5	71.0	3.57
Extended Summer	7.8	175.2	2.64
All Year	12.6	363.1	2.56
All Cottages	100.0 (N=1139)	116.03	3.07

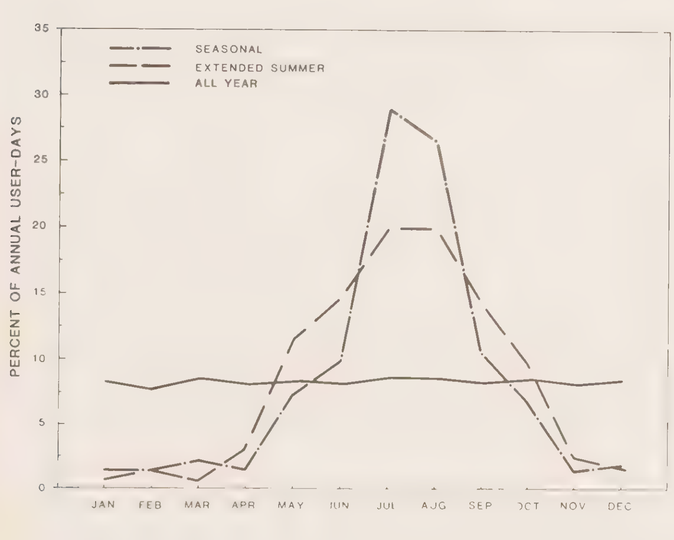


Figure 9. Patterns of cottage use, by cottage typology group

10.2 REGRESSION EQUATION

The regression equation for the Cottager Model was developed through multivariate analysis, by employing stepwise regression procedures to identify the best equation.

All the procedures produced the same solution from Steps 1 to 4, inclusive (Table 11). In Steps 5 and 6, all the procedures except Backward Elimination identified the same variables as explanatory predictors.

The coefficient of multiple determination increased with the addition of each variable to Step 5 ($R^2 = 0.920$) but gained little thereafter. The C_p approximated the number of variables in the six-variable equation more closely than the number in the five-variable one, indicating that the former could predict almost as well as an equation using all possible predictors (Table 12). The mean square error improved only slightly in Step 6.

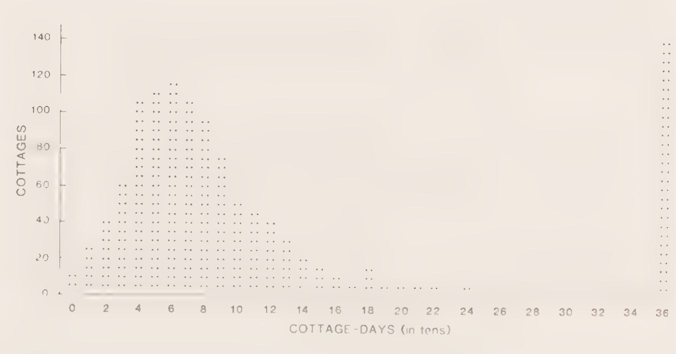


Figure 10. Frequency distribution of cottage-days

Table 11. Cottager Model regression equations identified by five stepwise procedures

Number of Variables in Equation	MIN R^2 and MAX R^2		STEPWISE		FORWARD		BACKWARD	
	R^2	Variable Added	R^2	Variable Added	R^2	Variable Added	R^2	Variable Dropped
1	0.838	PERM	0.838	PERM	0.838	PERM	0.838	PERM
2	0.911	ESUMMER	0.911	ESUMMER	0.911	ESUMMER	0.911	ESUMMER
3	0.918	TMODE1	0.918	TMODE1	0.918	TMODE1	0.918	TMODE1
4	0.919	TMODE2	0.919	TMODE2	0.919	TMODE2	0.919	TMODE2
5	0.920	DISTSUMH*	0.920	DISTSUMH*	0.920	DISTSUMH*	0.920	INTERACT
6	0.920	TMODE3	0.921	TMODE3	0.921	TMODE3	0.920	DISTSUML
7	0.921	DISTSUML INTERACT/ DISTSUMH	0.921	DISTSUML	0.921	DISTSUML	0.921	TMODE3
8	0.921	MAXPOORX	0.921	INTERACT	0.921	INTERACT	0.921	MAXPOORX
9	0.921	DISTSUMH	0.921	DISTSUMH			0.921	DISTSUMH

*Five variables in equation.
/ Variable dropped.
R^2 = Coefficient of Multiple Determination

Table 12. C_p and MSE for regression equations, Cottager Model

Steps	MIN R^2 and MAX R^2		STEPWISE		FORWARD		BACKWARD	
	C_p	MSE	C_p	MSE	C_p	MSE	C_p	MSE
1	1173.39	1708.96	1173.39	1708.96	1173.39	1708.96	1173.39	1708.96
2	129.71	935.64	129.71	935.64	129.71	935.64	129.71	935.64
3	44.57	871.84	44.57	871.84	44.57	871.84	44.57	871.84
4	20.67	853.38	20.67	853.38	20.67	853.38	20.67	853.38
5	11.41	845.77	11.41	845.77	11.41	845.77	15.50	848.81
6	8.67	842.99	8.67	842.99	8.67	842.99	9.38	843.52
7	6.42	840.58	10.01	843.25	10.01	843.25	6.41	840.58
8	8.01	841.01	8.42	841.32	8.42	841.32	8.01	841.01
9	10.00	841.75	6.42	840.58			10.00	841.75

MSE = Mean Square Error
C_p = Mallory's C_p

In view of the similarity at Steps 5 and 6 in the coefficient of multiple determination and the mean square error, the following five-variable equation was selected.

$$R_C = 68.312 + 278.578\ \text{PERM} + 102.042\ \text{ESUMMER} + 21.291\ \text{TMODE1} + 12.623\ \text{TMODE2} - 0.191\ \text{DISTSUMH}$$

where R_C = annual cottage-days per cottage predicted by the Cottager Model regression equation

PERM = proportion of cottages in the All Year cottage typology group on the benchmark lake

ESUMMER = proportion of cottages in the Extended Summer cottage typology group on the benchmark lake

TMODE1 = 1 when the travel mode is "car in summer, car in winter" (otherwise = 0)

TMODE2 = 1 when the travel mode is "car in summer, other in winter" (otherwise = 0)

DISTSUMH = the distance in kilometres from the cottage to the nearest high order urban centre

The All Year variable, which was entered first in the regression procedure, explained nearly 84% (R^2 = 0.838) of the variation in annual cottage-days. When Extended Summer was entered next, it explained a further 7% of the variation in cottage-days. Although all of the regression coefficients in the equation were highly significant, the variables representing cottage typology groups were the most reliable predictors.

The estimated regression function showed a remarkably good fit between the equation and the data (Table 13). The coefficient of multiple determination indicated that annual cottage-days per cottage were predictable with a

high degree of accuracy (R^2 = 0.920) from the linear combination of cottage typology and travel mode characteristics represented in the equation. Moreover, the close correspondence between the error sum of squares (SSE = 958261) and the prediction error sum of squares (PRESS = 969312) provided assurance that the equation should predict annual cottage-days nearly as accurately if applied to an alternative sample of cottages from the same region.

The ratio of the explained portion of the variation in annual cottage-days to the unexplained portion was extremely high (F = 2612.55). In addition, the high significance level ($p < .0001$) indicated that the regression was unlikely to be a product of chance.

The Student's t test was applied as each independent variable was entered into the equation. In each case, the hypothesis that no relationship existed between the dependent variable (annual cottage-days) and the independent variable was rejected.

Together, the five variables in the Cottager Equation explained 92% (R^2 = 0.920) of the variation in annual cottage-days among cottages in the survey sample. This was much higher than the 34% of variance (R^2 = 0.340) explained by the Accessibility Equation. However, when predicting for a specific lake, the benchmarking process dilutes the strength of the Cottager Equation. It is, therefore, more appropriate to compare the performance of the two models under conditions which simulate practical situations. The results of such tests are presented in Chapter 11.

The method of determining values for the travel mode variables and the distance to the nearest high order urban centre was the same for the Accessibility and Cottager Equations. The values for the cottage typology groups were derived from a benchmark lake, as described in the following subsection.

Table 13. Regression equation selected for Cottager Model

Annual Cottage-days = 68.312 + 278.578 PERM + 102.042 ESUMMER + 21.291 TMODE1 + 12.623 TMODE2 − 0.191 DISTSUMH

Variable	Regression Coefficient	Standard Error	Student's t	p
PERM	278.578	2.952	94.38	.0001
ESUMMER	102.042	3.234	31.51	.0001
TMODE1	21.291	2.199	9.68	.0008
TMODE2	12.623	2.322	5.44	.0001
DISTSUMH	−0.191	0.057	− 3.35	.0001

Coefficient of Multiple Determination (R^2)	= 0.920
Analysis-of-variance F-ratio (F)	= 2612.55
Error Sum of Squares (SSE)	= 958261
Predicted Error Sum of Squares (PRESS)	= 969312
Mallory's C_p (C_p)	= 11.41
Mean Square Error (MSE)	= 845.8

10.3 BENCHMARKING PROCEDURE

The benchmarking procedure was a crucial part of the Cottager Model, as the estimates of cottage typology proportions for the target lake (where development was proposed) were derived from a benchmark lake.

The logic and justification for benchmarking were discussed in the conceptual approach to the Cottager Model (Chapter 8). Criteria for selecting the most appropriate lake from the pool of candidates are outlined in this chapter.

Pool of Candidate Benchmark Lakes

The benchmark lakes were, by definition, lakes for which the cottage typology was known. Thus, the only lakes or portions of lakes which qualified were those included in the 1978 Survey of Lakeshore Residents. It was also essential that each benchmark lake have a sufficient number of cottages to allow each typology group to be represented. Accordingly, the pool of candidate benchmark lakes was limited to the 36 lakes with nine or more cottages.

The cottage and cottage-day proportions in each typology group, for each of the 36 benchmark lakes, are recorded in Table 14. The 1978 Survey of Lakeshore Residents provided a good cross-section of different cottage typology proportions in the Muskoka-Haliburton area. For example, while the cottage typology for the entire sample was 79.5% Seasonal, 7.8% Extended Summer, and 12.6% All Year, on Galla Lake all the cottages were in the Seasonal group. In contrast, on Utterson Lake only 40% were in the Seasonal Group while 55% were in the All Year group.

The pool of candidate benchmark lakes need not be limited to 36 lakes in the future. If cottagers are surveyed on other lakes and the responses are analyzed to obtain cottage typology proportions, these other surveyed lakes can be added as benchmark candidates.

Criteria for Selecting a Benchmark Lake

To select the most appropriate benchmark lake for a specific target lake, the goal was to find the optimum match in terms of cottage typology. As the single strongest predictor in the Cottager Model was the proportion of cottages in the All Year group, the best available indicators of the presence or absence of cottages in this group were employed. There were two possible approaches – one direct and the other indirect.

Direct Approach

The direct approach involved drawing on planning policies and zoning regulations as a source of information regarding permanent versus seasonal use of cottages in the future. Where a zoning by-law prohibited permanent use on the target lake, for example, this situation could be simulated on the benchmark lake by selecting a lake with no cottages in the All Year group.

Alternatively, where the proposed plan of subdivision was in a Limited Services area, it could be assumed that the

Table 14. Cottage and cottage-day proportions in each typology group, for 36 benchmark lakes grouped into seven clusters

Lake Name	COTTAGE PROPORTIONS			COTTAGE-DAY PROPORTIONS		
	Seasonal*	Extended Summer	All Year	Seasonal	Extended Summer	All Year
Cluster 1:						
Oudaze	.9231	.0000	.0769	.6500	.0000	.3500
Waseosa	.6666	.2222	.1110	.2780	.3635	.3586
Walker	.5999	.0667	.3333	.2874	.0748	.6378
Utterson	.4000	.0500	.5500	.1310	.0281	.8409
Muskoka 3	.7600	.1040	.1360	.4785	.1388	.3827
Muskoka 4 (TM1)	.7816	.1149	.1035	.4975	.1626	.3401
Muskoka 5 (TM2)	.9375	.0625	.0000	.8974	.1026	.0000
Muskoka 6 (TM3)	.8500	.1000	.0500	.6327	.1787	.1885
Cluster 2:						
Peninsula	.6666	.0000	.3333	.3130	.0000	.6870
Green	.3810	.0000	.6190	.1033	.0000	.8967
Pine	.3214	.0357	.6429	.0845	.0286	.8868
Kahshe	.9000	.1000	.0000	.7740	.2260	.0000
Canning	.9483	.0517	.0000	.9031	.0969	.0000
Kashagawigamog 1	.4763	.0476	.4762	.1465	.0502	.8034
Kashagawigamog 2	.6123	.1224	.2653	.2729	.1515	.5756
Gull	.9130	.0435	.0435	.7540	.0664	.1796
Cluster 3:						
Gaunt Bay	.6666	.1111	.2222	.3683	.1265	.5021
Muskoka 1 (TM1)	.7966	.1180	.0847	.5406	.1935	.2659
Muskoka 2 (TM2)	.8750	.1250	.0000	.8024	.1976	.0000
Lake of Bays	.7600	.0400	.2000	.4102	.0619	.5279
Dickie	.9032	.0323	.0645	.6992	.0751	.2257
Allen	.9444	.0000	.0556	.7792	.0000	.2208
Cluster 4:						
Galla	.9999	.0000	.0000	1.0000	.0000	.0000
Kawagama	1.0000	.0000	.0000	1.0000	.0000	.0000
Straggle	.9524	.0476	.0000	.8875	.1125	.0000
Kennaway	.9375	.0625	.0000	.8133	.1867	.0000
George's	.9000	.1000	.0000	.6988	.3012	.0000
Little Straggle	1.0000	.0000	.0000	1.0000	.0000	.0000
Cluster 5:						
East	1.0000	.0000	.0000	.9998	.0000	.0000
Cluster 6:						
Six Mile 1 (TM2)	.9143	.0857	.0000	.8467	.1532	.0000
Six Mile 2 (TM3)	.8981	.0926	.0093	.7440	.2100	.0460
Cluster 7:						
Little Kennisis	.8125	.1875	.0000	.6251	.3749	.0000
Haliburton	.6808	.1064	.2128	.3414	.1409	.5177
Riley 1 (TM1)	.8379	.0811	.0811	.5419	.1551	.3029
Riley 2 (TM2)	.7500	.1667	.0833	.4503	.2811	.2686
Riley 3 (TM3)	.9000	.1000	.0000	.7427	.2573	.0000

*Seasonal cottage proportion is not used in the Cottager Model equation. It is included in this table as the sum of the three cottage proportions is 1.

possibility of the road being closed in winter would deter permanent residents. Once again, this situation could be simulated by selecting a benchmark lake with no cottages in the All Year group.

Indirect Approach

An indirect approach was required when neither planning policies nor zoning by-laws were in effect or where permanent use was permitted. This approach relied on the type of access as an indicator of the annual level of cottage use. In other words, it drew on knowledge gained from the Accessibility Model.

Cottages in the All Year typology group had the highest annual level of cottage use – an average of 363 cottage-days. Their presence in this group was associated with certain accessibility characteristics. For example, the cottages tended to have access by car both in summer and in winter; to be served by a hard-surfaced road; and to be located near a high order urban service centre. Thus, indirectly, through similarities in accessibility features

Table 15. Accessibility characteristics of 36 benchmark lakes, grouped into seven clusters

Lake Name	Modal Travel Mode TM1	TM2	TM3	TM4*	Mean MAXPOOR**	Mean DISTSUMH***
Cluster 1:						
Oudaze	0	1	0	0	5.3	22.4
Waseosa	1	0	0	0	7.1	15.5
Walker	1	0	0	0	1.0	16.2
Utterson	1	0	0	0	0.5	19.0
Muskoka 3	1	0	0	0	1.0	20.7
Muskoka 4 (TM1)	1	0	0	0	0.9	13.5
Muskoka 5 (TM2)	0	1	0	0	0.9	14.2
Muskoka 6 (TM3)	0	0	1	0	1.2	13.4
Cluster 2:						
Peninsula	1	0	0	0	0.4	12.0
Green	1	0	0	0	0.6	13.6
Pine	1	0	0	0	0.6	12.9
Kahshe	0	0	1	0	1.6	12.6
Canning	1	0	0	0	1.5	10.0
Kashagawigamog 1	1	0	0	0	4.2	12.5
Kashagawigamog 2	1	0	0	0	0.4	11.6
Gull	1	0	0	0	0.8	6.8
Cluster 3:						
Gaunt Bay	1	0	0	0	1.0	31.5
Muskoka 1 (TM1)	1	0	0	0	2.4	28.9
Muskoka 2 (TM2)	0	1	0	0	3.1	28.8
Lake of Bays	1	0	0	0	0.3	25.6
Dickie	1	0	0	0	1.2	28.0
Allen	0	1	0	0	13.3	41.0
Cluster 4:						
Galla	0	1	0	0	2.6	58.2
Kawagama	0	1	0	0	6.2	49.2
Straggle	0	1	0	0	5.4	56.1
Kennaway	0	1	0	0	2.9	53.5
George's	0	1	0	0	5.4	56.0
Little Straggle	0	1	0	0	5.8	56.4
Cluster 5:						
East	1	0	0	0	15.5	60.6
Cluster 6:						
Six Mile 1 (TM2)	0	1	0	0	4.4	47.2
Six Mile 2 (TM3)	0	0	1	0	4.5	44.7
Cluster 7:						
Little Kennisis	0	1	0	0	6.5	37.1
Haliburton	1	0	0	0	2.2	33.3
Riley 1 (TM1)	1	0	0	0	1.1	41.9
Riley 2 (TM2)	0	1	0	0	2.6	42.5
Riley 3 (TM3)	0	0	1	0	1.0	42.2

*TM1 = Travel Mode 1 (car in summer, car in winter)
TM2 = Travel Mode 2 (car in summer, other in winter)
TM3 = Travel Mode 3 (boat in summer, other in winter)
TM4 = Travel Mode 4 (car or boat in summer, no winter use)
**MAXPOOR = distance on poor roads to reach the cottage
***DISTSUMH = distance from the cottage to the nearest high order urban centre

such as these, it was possible to match the target lake with a benchmark lake that had a similar proportion of cottages in the All Year group (Table 15).

In selecting a benchmark lake, several objective criteria can be applied but there is still an element of subjective judgment inherent in the process. This judgment will normally be exercised by knowledgeable planners and others who have a degree of expert opinion regarding the relevant characteristics of the lakes in question. Thus, the need for skilled judgment in the selection of a benchmark lake can be considered a strength rather than a weakness of the benchmarking system.

When benchmarking is introduced, the theoretical power of the Cottager Model is weakened. Nevertheless, it is evident from the tests of performance described in Chapter 11 that the model retains sufficient capability to qualify as a practical predictive tool.

10.4 PREDICTED COTTAGE-DAYS FOR SUBDIVISION

The regression equation is applied to predict annual cottage-days per cottage. As the prediction is based on the patterns of cottage use identified in the cottage typology, it can be applied to the entire subdivision.

The assumption regarding the number of cottages in the proposed subdivision is the same as for the Accessibility Model, that is, each lot represents one potential future cottage.

To apply the regression equation to a given subdivision, the procedure is to multiply the predicted annual number of cottage-days per cottage by the number of lots in the subdivision. For example, if there are 90 lots in the subdivision and if 200 cottage-days per cottage are predicted by the regression equation, the total annual number of cottage-days can be calculated by multiplying 90 x 200 which equals 18,000 cottage-days.

10.5 CONVERSION TO USER-DAYS

The method of converting predicted cottage-days to user-days is more complicated for the Cottager Model than for the Accessibility Model, as it takes into account differences among the cottage groups in both average annual cottage use and average household size. These differences were evident in Table 9.

In applying the regression equation to predict cottage-days, a benchmark lake was selected to provide the proportion of cottages in each of the three cottage groups. The same benchmark lake is used in the conversion process to provide the proportion of cottage-days attributable to each cottage group (Figure 7).

The effects of differences in the annual level of cottage use from one typology group to another are illustrated in Figure 11, which shows the distribution of cottages and cottage-days, by typology group. For the All Year group, for example, it is evident that the percentage of cottage-days is much higher than the percentage of cottages, due to relatively high annual use per cottage. Conversely, although the Summer, Weekends and Long Weekends groups together comprise 80.1% of all cottages, they generate only 48.6% (Seasonal group) of cottage-days.

The method of converting cottage-days to user-days, for the subdivision, involves the following steps:

Step A. Allot the total cottage-days predicted by the Cottager Model equation for a proposed subdivision to each of the three cottage groups, using cottage-day proportions from the benchmark lake as the basis for allotment.

Step B. Multiply the cottage-days for each cottage group by the average household size for that group, to obtain total user-days for each group.

Average Household Size

Seasonal	3.57 persons per cottage
Extended Summer	2.64 persons per cottage
All Year	2.56 persons per cottage

Step C. Total the user-days for the three groups, to produce predicted user-days for the entire subdivision.

An example of this method of converting cottage-days to user-days is provided in Table 16.

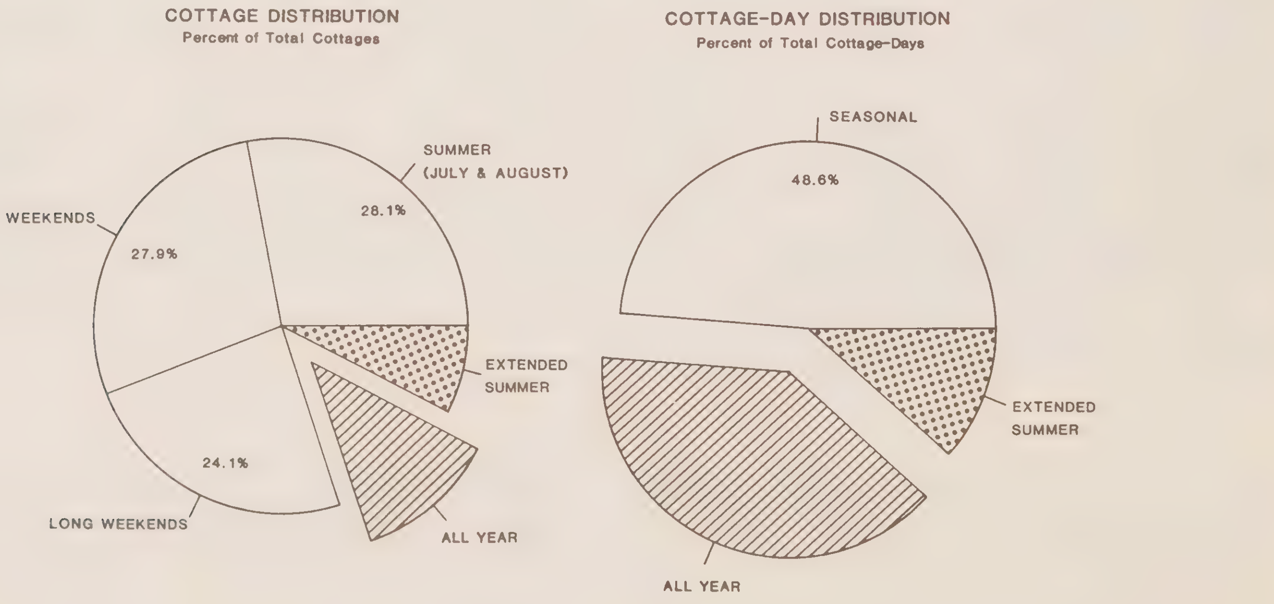


Figure 11. Cottage and cottage-day distribution among cottage typology groups

Table 16. An example of the method of converting cottage-days to user-days

A	B	C	D	E	F	G
Predicted Cottage-days for Subdivision	Cottage Typology Group	Proportion of Cottage-days	Number of Cottage-days	Number of Persons per Cottage	Number of User-days	Predicted User-days for Subdivision
.18,000	SEASONAL	0.60	10,800	3.57	38,556	57,132
	EXTENDED SUMMER	0.10	1,800	2.64	4,752	
	ALL YEAR	0.30	5,400	2.56	13,824	

NOTES
Column A. Assumed. In practice, this would be predicted by applying the regression equations.
Column C. Assumed. In practice, these proportions would be derived from the benchmark lake.
Column D. Calculated by multiplying the predicted cottage-days (A) by the proportion of cottage-days (C) in each typology group (B).
Column E. Average number of persons per cottage, computed from data obtained from cottagers through the 1978 Survey of Lakeshore Residents (Table 9).
Column F. Calculated for each typology group by multiplying the number of cottage-days (D) by the number of persons per cottage (E).
Column G. Calculated by aggregating the number of user-days for each typology group (F).

10.6 THE MODEL

The network diagram depicting the conceptual approach to the Cottager Model (Figure 7) provided an overview of the key aspects of the model which have been described in this chapter.

The Cottager Model can also be expressed in the form of an equation:

PREDICTED ANNUAL USER-DAYS FOR SUBDIVISION = $(R_C * N) * [(S_c * S_x) + (E_c * E_x) + (A_c * A_x)]$

where R_C = annual cottage-days per cottage predicted by Cottager Model regression equation

N = number of lots in subdivision

S_c = proportion of cottage-days in Seasonal group of cottages on benchmark lake

S_x = average number of persons per cottage (3.57) for Seasonal group of cottages

E_c = proportion of cottage-days in Extended Summer group of cottages on benchmark lake

E_x = average number of persons per cottage (2.64) for Extended Summer group of cottages

A_c = proportion of cottage-days in All Year group of cottages on benchmark lake

A_x = average number of persons per cottage (2.56) for All Year group of cottages

11. ACCURACY OF MODEL PREDICTIONS

The regression equations for the Accessibility Model and the Cottager Model were evaluated in Chapters 9 and 10, respectively, in terms of their ability to explain the variation in cottage use among the 1139 cottages in the survey sample. It was important also to establish their accuracy at the level of discrete lakes, since the models were designed to predict cottage use for a proposed subdivision on a specific lake. Accordingly, their predictions were tested on each of the 36 benchmark lakes.

The benchmark lakes were employed for test purposes because survey data were available recording cottage use. This survey information represented "actual" cottage use.

The first step in testing prediction accuracy was to compare the actual and predicted values in terms of a percentage difference, i.e. the difference between the two figures was calculated as a percentage of the actual value. These percentages indicated how closely the model predictions resembled the cottage use levels actually observed. A low percentage difference signified a high degree of accuracy; a positive difference indicated over-prediction; a negative difference indicated under-prediction.

The second step was to establish categories representing the cumulative percentage difference between predicted and actual cottage use. These categories were "within 10% of actual", within 20%, within 30%, and so on. Each of the 36 lakes was then assigned to the appropriate category. Finally, the percentage of the lakes in each category was calculated.

The third step was to compare, for each category, the predictions from the Accessibility Model and the Cottager Model with those from the Averaging Model. The latter prediction of user-days was calculated by multiplying the number of cottages on the lake by the average number of cottage-days (116.03) annually, by the average household size (3.07 persons per cottage).

If the Accessibility Model and Cottager Model were to have practical value in capturing variations in the level of cottage use, their predictions would need to be more accurate than those produced by the Averaging Model.

11.1 ACCESSIBILITY MODEL

When the Accessibility Model was applied to predict cottage use, the predominant accessibility characteristics for all cottages on the lake were substituted for exact data. This was done to simulate the level of detail that would be possible when the model was used in practice.

For the travel mode variables, the imputed values were derived from the most typical travel mode reported by survey respondents for cottages on each lake. For example, if 80% of the cottages on a specific lake were accessible by car all year, the "imputed" travel mode for all cottages on the lake would be car all year. For the continuous variables, distance to the nearest high order urban centre and distance on poor roads, the imputed values were the mean distances for all cottages on the lake.

Expressed in units of annual user-days, predictions from the Accessibility Model were within 20% of actual user-days for 60% of the lakes, while predictions based on the Averaging Model reached this level of accuracy for only 35% of the lakes (Figure 12). Similarly, nearly 70% of the Accessibility Model predictions were within 30% of actual user-days, while the Averaging Model predictions achieved this level of accuracy for less than 50% of the lakes.

These comparisons indicated that the Accessibility Model, by taking into account differences in cottage access, provided more accurate predictions of cottage use than the Averaging Model.

One of the reasons why the Accessibility Model predicted as well as it did was that permanent residents were identified indirectly through the travel mode. This is evident in Figure 13, which indicates that almost all cottages used all year are accessible by car in summer and winter.

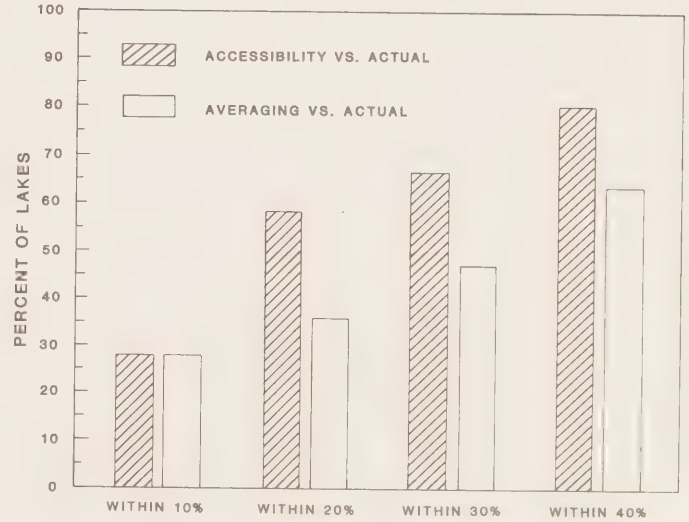


Figure 12. Accessibility Model (imputed data): predicted user-days compared with actual user-days

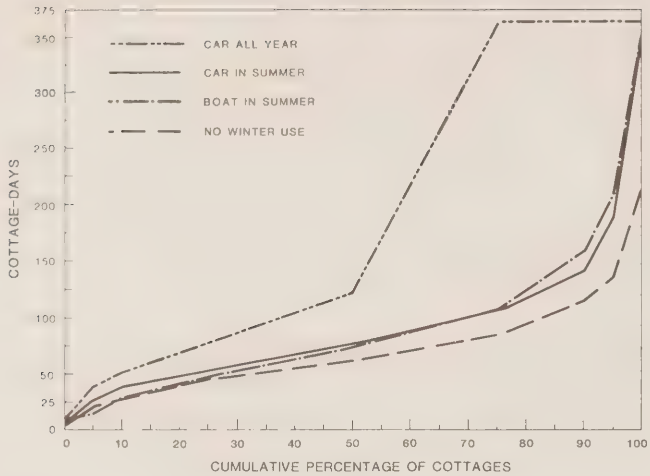


Figure 13. Frequency distribution of travel modes and cottage-days

11.2 COTTAGER MODEL

When the Cottager Model is applied to predict cottage use, the cottage typology proportions are normally derived from a benchmark lake. However, a situation could occur where a new subdivision is proposed on one of the benchmark lakes. In such a case, it would be more appropriate to use the known cottage typology for existing cottages on the same lake, rather than a different benchmark lake.

A somewhat similar situation occurred when the Cottager Model was tested, because the testing was done on the 36 benchmark lakes. In this case, however, a valid test demanded that the performance of the model be evaluated on the basis of the way it would normally be applied. Therefore, for test purposes, the cottage and cottage-day typology proportions had to be derived from a different benchmark lake.

The Cottager Model predictions were within 10% of actual cottage use for about 30% of the lakes; within 20% for about 65% of the lakes; and within 30% for 75% of the lakes (Figure 14). These distributions provided a

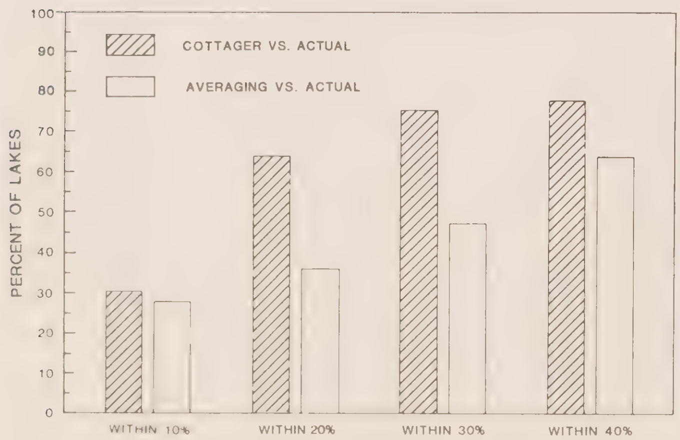


Figure 14. Cottager Model (benchmark data): predicted user-days compared with actual user-days

fairly detailed evaluation of the level of accuracy attained by the model under conditions similar to those in practice.

The Cottager Model predicted within 20% and within 30% of actual cottage use for a substantially higher percentage of lakes than the Averaging Model. For example, the Cottager Model predictions were within 20% of actual for over 60% of the lakes, while the Averaging Model achieved this level for less than 40% of the lakes.

These comparisons demonstrated that the Cottager Model, by taking into account differences in the pattern of cottage use, provided predictions that were generally more accurate than those produced by the Averaging Model.

11.3 EVALUATION AND CONCLUSIONS

The tests of prediction accuracy at the level of discrete lakes provided a basis for evaluating the performance of each model, with a view to deciding whether to retain one or both.

The Accessibility Model predicted better than the Averaging Model, even when imputed data were employed to simulate the degree of precision possible in practice.

The model was objective, simple and efficient – all desirable features. It was objective in the sense that all variables could be measured; relatively simple to apply because it employed only five predictor variables; and efficient in achieving good results with a minimum of effort.

A major advantage of the Accessibility Model was that its predictions could reflect future conditions to some extent. This was possible because Official Plans and Transportation Plans would be used as sources of information for coding the variables. Thus, proposed new roads, road improvements and realignments would be taken into account to the degree to which they could be predicted when the model was applied. Such improvements in accessibility would be reflected in the predictions of cottage use. Therefore, they would tend to be more realistic than predictions based solely on the existing transportation network.

The Cottager Model predicted user-days better than the Averaging Model, even when benchmark data and imputed data were employed to simulate application of the model in practice.

The Cottager Model was more complicated than the Accessibility Model but the logic of the method was clear and valid at each step. Moreover, as the model was programmed in the computer, the calculations did not prove onerous for the user. Thus, the Cottager Model offered a feasible method of predicting.

The benchmarking procedure in the Cottager Model involved an element of subjective judgement. However, this would normally be exercised by knowledgeable planners, or others with some familiarity with local conditions, so it could be considered a strength rather

than a weakness of the benchmarking. In any event, the model proved to be capable of generating predictions that were sufficiently accurate to make it a practical predictive tool.

The test results offered no statistical support for choosing one model over the other. The predictive power of the Cottager Model in practice appeared to be in a range similar to that of the Accessibility Model. Therefore, both models should be retained.

When the Accessibility Model and the Cottager Model are incorporated into the Ontario Lakeshore Capacity Simulation Model (OLCSM), one prediction of cottage use must be selected as input to the Water Quality Submodel and the Fisheries Submodel. Accordingly, a method of choosing the appropriate prediction is outlined in Chapter 12.

The Cottager Model has some potential for improvement. The greatest opportunity lies in the use of planning policies relating to seasonal versus permanent use of cottages in the benchmarking procedure. Where a local government has a policy prohibiting permanent use in designated areas and where it has a zoning by-law indicating where the policy applies, the cottage typology for any proposed subdivision affected by the prohibition can be assumed to exclude permanent residents. Such information can be applied directly in selecting a benchmark lake, by choosing a lake with no cottages in the All Year group. The effect of such a choice on the predicted number of annual user-days would be substantial, because cottages in the All Year group log the highest number of cottage-days (363 annually).

In testing the model predictions in the Study Area, it was not possible to incorporate planning policies regarding seasonal versus permanent use because the lakes used for test purposes were not subject to planning policies of this nature. For this reason, planning policies are not reflected in the evaluation of model performance.

Policies relating to seasonal versus permanent use are in force in some other parts of Ontario, so improvement in the predictive capability of the Cottager Model might be possible if the Lakeshore Capacity Study models were adapted to other areas. Similarly, if local governments in the Muskoka-Haliburton Study Area were to introduce such policies, the Cottager Model might be capable of producing better predictions there.

12. PRACTICAL APPLICATION OF THE LAND USE COMPONENT MODELS

The specific purpose of the Land Use Component models was to provide an estimate of future cottage development, expressed in units which could be used in the Lakeshore Capacity Study models. The Trophic Status model, for example, required an estimate of annual user-days and the Fisheries Net Productivity model required an estimate of cottage-days. When the various models developed by the individual components were integrated into the Ontario Lakeshore Capacity Simulation Model, predictions from the Land Use Component models became the driving variables.

The practical application of the Land Use Component predictive models is described in this section.

12.1 PREDICTING COTTAGE USE

The Accessibility Model and the Cottager Model were developed by the Land Use Component team to predict the amount of cottage use likely to be generated by a proposed development. When the accuracy of the models was tested, by comparing the model predictions with actual survey data, there was no statistical support for choosing one model over the other. Accordingly, both models should be used.

Thus, the first step is to generate two predictions of cottage use, by applying the Accessibility Model and the Cottager Model. The step-by-step procedure is set out in the User's Guide for the Ontario Lakeshore Capacity Simulation Model (Teleki and Herskowitz, 1982).

12.2 SELECTING THE PREDICTION

The OLCSM should be run with each prediction, to determine the impact on the environment that each would produce. In some cases, the environmental impact will not be sufficient to cause deterioration, so a choice between the cottage-use predictions will not be needed. In other cases, the environmental impact resulting from the higher and lower cottage-use predictions might be substantially different. In such situations, a method of selecting the appropriate cottage-use prediction will be necessary.

With this in mind, various analyses were undertaken in an attempt to determine where each model predicted with greater accuracy, but there were no conclusive results. Also explored was the question of whether the high or low prediction, or their midpoint, was the most accurate. It was evident that the high prediction most closely approached the actual (survey) data where the travel mode was "car all year" and where the proportion of cottages in the All Year group was relatively high. However, the analysis offered no clear directive to assist in selecting the best prediction in other situations.

Consideration was then given to the environmental implications of using a prediction that was too low or too high. If the prediction were too low, the predicted environmental impact would also be too low. Based on this impact, the decision might be to allow more cottages than the environment could actually support. Thus, a low prediction of cottage use could result in damage to the environment.

Conversely, if the predicted cottage use were too high, the predicted environmental impact would also be too high. Based on this impact, the decision might be to allow fewer cottages than the environment could actually support. However, the environment would be protected and it would still be possible to add cottages later if, after monitoring actual change, this proved to be a safe course of action.

In view of these implications for the environment and the lack of clear statistical directives, the higher prediction of cottage use should always be chosen.

12.3 COTTAGE CONVERSION

As the process of "cottage conversion" from seasonal to permanent use is familiar to planners, it may be useful to explain how this process is taken into account in the predictive models.

It was decided at the outset to collect data on actual cottage use, to provide a data base for developing the predictive models. Accordingly, each cottage household in the sample was asked to record actual use of the cottage over a period of twelve months. Cottages used all year therefore included two types: (1) those constructed initially for permanent use, and (2) those converted from seasonal to permanent use. It is the presence of the latter – the converted cottages – which is important in understanding the nature of the model predictions.

As the models were developed from the survey data, the predictions generated by the models take into account the greater number of cottage-days per year associated with converted (and other) permanent cottages.

Analysis of the 1978 Survey of Lakeshore Residents data indicated that the net rate of conversion from seasonal to permanent use was extremely low in the Study Area. This

suggests the possibility that conversions from permanent to seasonal use also occur and may be sufficient to largely offset conversions from seasonal to permanent. In order to obtain more precise information relating to cottage conversion, it is recommended that conversion both to and from permanent use be explored specifically in any survey of cottages that may be undertaken in the future.

12.4 KEEPING THE MODELS CURRENT

The model predictions reflect the 1978 Survey of Lakeshore Residents data on cottage use, from which they were developed. Although changes may occur slowly, over the years changes can be anticipated in the annual amount of cottage use. Some of the factors associated with such change are fuel costs for cars and boats, the level of disposable income, the amount of leisure time available, and family age and composition. While a change in any one of these factors may bring about some change in cottage use, there is no way of predicting how quickly cottagers may react.

There are also strong stabilizing influences which tend to balance the pressures for change. For example, both the school summer vacation period and the climate encourage highest cottage use during July and August.

A new survey of cottages may not be needed for five or ten years, depending on the rate of change. When the Land Use Component data from cottage surveys in 1976 and 1978 were compared, for example, the levels of cottage use were found to be similar over this two-year period. It is recommended that a limited number of lakes that were surveyed in 1978 be monitored periodically in the future, to determine how much change has occurred since 1978. In this way, the pace of change can be determined and the need for a broader survey can be anticipated.

Now that predictive models have been developed and their information requirements established, the time and work involved in a new survey would be substantially less than that required for the original Land Use Component work.

12.5 ADAPTING THE MODELS TO OTHER PARTS OF ONTARIO

The Land Use Component models, which apply at present to the Muskoka-Haliburton Study Area, can be adapted for use elsewhere, wherever data are available regarding cottage use levels and patterns.

The work involved in data collection and analysis would be substantially less than in the original Study Area, now that models are in place and their data needs are known. However, the seasonal use of cottages restricts surveys to July and August, so the preparation stage for the survey work must adhere to a rigid time schedule.

12.6 REFINING THE MODELS IN THE FUTURE

It may be possible in the future to refine the Land Use Component models or to simplify their application. Initially, however, it is strongly recommended that the Land Use Component models be applied in their present form, in accordance with the procedures described in this report.

Particular care must be taken in designating the "travel mode" correctly, as it affects not only the predictive power of the Accessibility Model but also the accuracy of the benchmarking which, in turn, affects the predictive power of the Cottager Model.

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LAKESHORE CAPACITY STUDY

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A Simulation Model for Predicting the Impact of Lakeshore Cottages on the Environment

JULY 1986

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Wildlife
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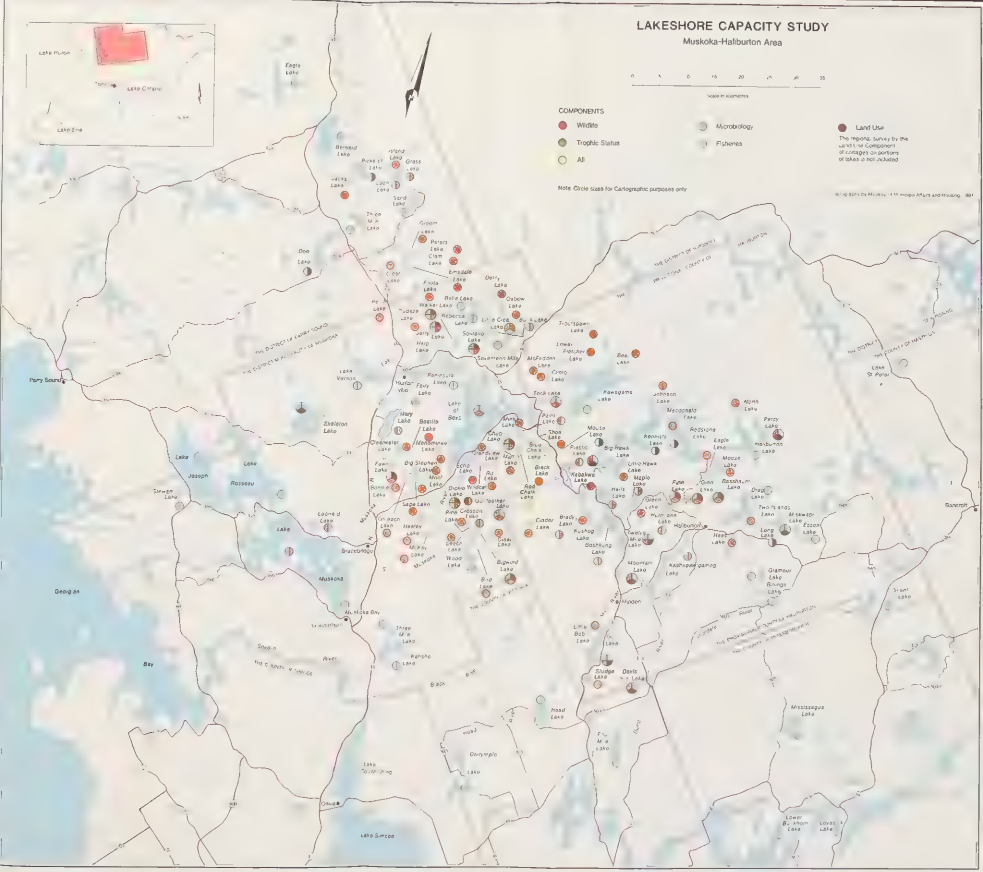
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FOREWORD

Community planners and other professionals involved in the preparation of planning policies for lakeshore development and in the review of specific subdivision proposals have always found it difficult to determine objectively the impact of development on the natural environment. In response to this challenge, the Lakeshore Capacity Study was undertaken to provide a planning tool to assist in evaluating the effects of cottage development on inland lakes and lakeshores. Central to the task was the need to gain a clearer understanding of the relationship between cottage development and its impacts on selected aspects of the natural environment.

To accomplish these objectives, the Ministry of Municipal Affairs carried out the Lakeshore Capacity Study in co-operation with the Ministry of the Environment and the Ministry of Natural Resources.

The Muskoka-Haliburton area of central Ontario was chosen as the Study Area. The homogeneity of the area, which is part of one physiographic region, reduced the need to account for major natural variations among the lakes and watersheds. In addition, the extent of existing development on the lakes varied, permitting an examination of situations ranging from no development to full development.

The Study involved measuring the source of environmental impact, in terms of the lakeshore cottages and their use, and the impact of cottage development in terms of water quality, fisheries, and wildlife habitat. The methods of prediction derived from the research were linked in a simulation model, which is capable of predicting trends in the values of the key indicators of impact.

As the research, analysis and findings of the Study are documented in a set of seven reports, selective reading may be desirable. Those readers who have a general interest in the work are advised to read the Committee Report first, as it provides an overall summary of the findings. This should be followed by the Integration report, in which the simulation model is described. Readers with more specialized interests will find the details of each component of the Study in the other five reports. In each report, the reader can select from the table of contents the most important chapters for his or her purposes.

The end product of the Study, the Ontario Lakeshore Capacity Simulation Model, has several features worth noting. The spatial unit addressed by the model is a single lake and the lakeshore. When the model is applied, the number of unknowns related to the natural environment can be substantially reduced, making it easier for planners or other professionals to weigh the environmental effects of development. The model goes a step further to permit predictions of the impact of cottage development when different management policies are selected.

The scope of the simulation model demands some explanation. In its present form, the model applies to cottage development on inland lakes in the Study Area, where the research was conducted. However, the methods of prediction can be adapted to other parts of the province, as long as differences in conditions are taken into account.

The purpose of the Study was to measure the environmental impact of cottages. Commercial and industrial uses were excluded deliberately, in order to simplify the difficult task of measuring cottage impact. The flexibility inherent in the simulation model makes it possible to add other types of land use later, if so desired.

Most of the existing cottage development in the Study Area is located in a single tier along the shoreline. For this reason, the simulation model applies to the immediate lakeshore and not to backshore development. Again, the methods of prediction developed for cottages near the lake can be adapted to measure the impact of cottage development in other forms.

The model was designed to measure the physical and chemical impacts of cottages. Accordingly, it does not address other planning concerns, such as social and economic impacts. While these were recognized as essential considerations in decision-making, the specific objective of Phase III of the Lakeshore Capacity Study was to find practical ways of producing better technical data regarding environmental impact.

Now that Phase III of the Study is completed, with the production of the Ontario Lakeshore Capacity Simulation Model (OLCSM), the next step envisaged is to apply the model experimentally within the Ministry. In this setting, model output can be tested in a variety of actual development situations. When this period of experimental use has been concluded and the results assessed, the three participating ministries will be able to determine whether the model should be adapted to other parts of the province and whether it should be made available more widely.

The Ministry of Municipal Affairs considers the OLCSM to be a potential planning tool but recognizes that the technical and organizational implications of its use must be examined. While this is underway, the model will be available for testing as an additional planning tool to supplement the information normally required to evaluate a planning policy or development proposal. However, the model will not be used in the decision-making process, which will still rest on the customary range of planning considerations.

This Integration report describes the Ontario Lakeshore Capacity Simulation Model (OLCSM) developed in Phase III of the Lakeshore Capacity Study.

ACKNOWLEDGEMENTS

Without the participation and support of the Lakeshore Capacity Study component managers and staff, the simulation model would not have been possible. For their many contributions in terms of original component research results and ideas, we thank C. Allan Burger, Microbiology; Peter J. Dillon, Trophic Status; Jean C. Downing, Land Use; David L. Euler, Wildlife; and Allen M. McCombie, Fisheries.

The fundamental structure of the simulation model was developed at two workshops, attended by over forty participants from three ministries. This model is, at least in part, the result of their creative efforts.

Finally, the assistance of ESSA Ltd. in the use of the Adaptive Environmental Assessment and Management approach, gave clear direction to the modelling process. We thank ESSA's Robert R. Everitt, Peter MacNamee, David R. Marmorek, Nicholas C. Sonntag and Michael J. Staley for designing and coding the initial model, and Pille Bunnell for preparing a comprehensive set of graphics to illustrate the model's development, internal structure and output.

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1. SUMMARY

The Ontario Lakeshore Capacity Simulation Model (OLCSM) was developed as part of the Lakeshore Capacity Study to assist in planning for cottage development on the shores of inland lakes in Ontario. With the model, it is possible to predict changes through time that are likely to occur in the land/lake environment, with respect to water quality, fisheries and wildlife, in response to shoreland cottage development.

The inner workings of the model are mathematical relationships, based on the current understanding of lake-watershed ecosystems. The model output, in the form of projections of selected environmental indicators, provides information on the development capacity of the lake under consideration.

In addition, different management scenarios can be simulated to determine the environmental consequences under each set of conditions. The various management options can then be weighed, based on tangible measures of environmental impact. With this type of analysis, several acceptable levels of cottage development may be predicted, depending on the management strategy selected.

1.1 THE APPROACH

The Adaptive Environmental Assessment and Management methodology, used to construct the model, prescribes a coordinated interdisciplinary approach that takes into account the needs and concerns of the ultimate users. Inherent in the process are short, intensive, modelling workshops which, for the Lakeshore Capacity Study, included scientists, planners, resource managers and policy advisors from the three participating ministries.

The OLCSM incorporates (a) the predictive models developed by researchers in the various components of the Lakeshore Capacity Study, (b) relevant data from the scientific literature, and (c) an intuitive understanding of the lake-watershed system. The present version of the model applies to the Muskoka-Haliburton area of central Ontario, where the research was conducted.

1.2 THE SUBMODELS

The OLCSM is divided into four subsystems, or submodels: Land Use, the source of the impact; and Water Quality, Fisheries and Wildlife-Habitat, the submodels that calculate the environmental impact. Cottage use, identified as the major source of environmental impact from lakeshore development, is predicted in the Land Use Submodel. Output from this submodel is used in the Water Quality, Fisheries and Wildlife-Habitat submodels.

The Water Quality Submodel has two parts: one predicts the lake nutrient level, or trophic status, plus related water quality indicators, such as oxygen levels and transparency; the other is concerned with the health-related microbiology of lake water.

The Fisheries Submodel calculates the amount of fishing pressure, harvest and fish stock in a lake, in response to changes in the number of cottages, cottage use and the number of anglers. The two sport fish species selected for inclusion in the model are lake trout and smallmouth bass. The submodel is designed so that the lake trout are affected by reduced oxygen concentrations in the lake, whereas the smallmouth bass population changes in response to habitat alterations in the littoral zone.

In the Wildlife-Habitat Submodel, the impact of cottage development on wildlife is calculated indirectly from changes in the shoreland vegetation. This "habitat approach" emphasizes wildlife habitat disturbance. The wildlife included in the model are small mammals, songbirds, mink, loons, hawks, deer and turtles.

The linkages between the submodels represent only the key interactions necessary for the prediction of impact.

1.3 INPUT, OUTPUT AND OPERATION

The submodel equations are designed to use data that are readily available. The input data necessary to run the model can be assembled largely from existing data bases. The input consists of lake-specific data, such as the lake morphometry, the lakeshore forest and littoral zone types around the lake, and the location of existing cottages, existing vacant lots and proposed cottage lots.

The output consists of values for a set of indicator variables which function as gauges, or measures, of the response of the system. These values are plotted as graphs, or printed in tabular form, for a user-specified interval of time and length of simulation. When alternative management strategies are simulated, the indicator values may be compared.

The management strategies correspond to the controls which can be exercised as part of a policy. The strategies can be introduced or terminated at any time during the simulation.

1.4 FUTURE REFINEMENTS

The OLCSM is not intended to be immutable. The present version is capable of predicting the environmental impact of lakeshore cottage development and testing the effects of various mitigative measures. The next version should incorporate some or all of the refinements cited in the report as desirable to improve the model's predictive capability. The model should then be tested in extreme situations, to ensure that it is responsive to the full range of conditions in the Study Area. Subsequent versions of

the OLCSM should reflect advances in the scientific understanding of lake ecosystems.

2. LAKESHORE CAPACITY STUDY CHRONOLOGY

2.1 PROBLEM DEFINITION

The objective of the Lakeshore Capacity Study (LCS) was to develop a comprehensive management and planning tool for assessing the impact of cottage development on inland lakes in Ontario. This approach was based on the premise that the preferred course in the management of a renewable resource is to avoid deterioration of the resource rather than to apply corrective measures after problems arise. The government recognized the need for a scientifically-based, systematic approach to the prediction of the impact of cottage development that would aid in the evaluation of proposed plans of subdivision, the preparation of official plans and the formulation of lake development policies.

2.2 RESEARCH PHASES — FORMATION OF LCS STRUCTURE

Phase I of the LCS was a feasibility study conducted by the Institute of Environmental Sciences and Engineering, University of Toronto (University of Toronto 1971). It consisted of a survey of literature outlining some of the major approaches to ecosystem modelling and a proposal for developing methods of measuring the impact of cottages on inland lakes in Ontario.

Phase II (University of Toronto 1974) included an elaboration of the recommended approach to the problem; the identification of ecological subsystems and subsystem elements that were important as indicators of the state of the system; an investigation of the adequacy of existing technical data for use in a predictive model; and an evaluation of the suitability of various physiographic regions in Ontario for field work. The Muskoka-Haliburton area was recommended as the most suitable location for conducting the research program (Figure 2.1). This area was selected as the lakes had similar geomorphological characteristics and a relatively uniform climate, thereby minimizing variation in the environmental parameters. In addition, lakes in the region had different levels of cottage development and environmental impact which was essential in order to establish a range of environ-

Figure 2.1 Location of Study Area

mental responses.

As outlined in Phase II, the system designated for study was the lake and its surrounding watershed. Two sets of variables were imposed on the system. The first was a set of natural forces such as solar radiation, precipitation, and wind, acting continuously on the system. The second was a set of anthropogenic (i.e. human-induced) forces, including cottaging, boating, swimming and angling, acting on the lake-watershed system at variable rates over a period of time.

The Phase III in-depth investigations were conducted by the Ontario ministries of Municipal Affairs and Housing, the Environment and Natural Resources. An interdisciplinary team of researchers from the three ministries was formed to develop predictive models with respect to cottage development and the direction and magnitude of environmental changes resulting from this type of development.

In the LCS Phase III, five research components were identified: Land Use, Trophic Status, Microbiology, Fisheries and Wildlife. Each represented a distinct part of the lake-watershed system and each involved research activities requiring specific technical knowledge. Within each component, important elements were chosen as indicators of the state of the subsystem.

The researchers conducted their investigations within the framework of the component structure. Because there was little existing research directly applicable to the Study objectives and substantial gaps in the scientific understanding of the resilience of the lake ecosystem to the impacts of development, the LCS investigators had to devise new approaches and research methods. The field data were used to generate quantitative relationships between development and the affected environmental parameters. For each component, a practical requirement was that the predictive capability be achieved principally with existing data, thus reducing the time expended in the review process.

It was anticipated that these research findings would then form the backbone of an integrated predictive model, capable of providing sound technical data to assist in the evaluation of proposed plans of subdivision and the preparation of official plans. The model was also expected to contribute to resource planning and management.

The impact of lakeshore cottages on the natural environment was the focus in Phase III. Human behaviour, attitudes, perceptions and socio-economic indicators were not explored. However, some aspects of the behavioural response to socio-economic factors were implicit in the methods of calculating cottage use and fishing effort and to that extent they were represented in the investigation.

2.3 LCS SUBSYSTEMS AND INDICATORS

The LCS components conducted research to derive methods of measuring and subsequently predicting the environmental changes likely to result from lakeshore cottage development.

The Land Use Component team investigated the source of the impact, which involved the cottages and cottagers, the extent and seasonal pattern of cottage use, and selected recreational activities in which cottagers participate. The relevant indicators for predictive purposes were the "cottage-day", one cottage in use for one day; and the "user-day", one cottage in use for one day by one person. Each can be expressed as a monthly, seasonal or annual value for all the lakeshore cottages on a given lake.

The Trophic Status Component group investigated the linkage between the man-made nutrient additions to a lake and its water quality. Lake phosphorus concentrations were used as the prime indicator of nutrient enrichment, as phosphorus is the nutrient most frequently controlling algae and aquatic plant growth in north temperate lakes. Other water quality indicators that are a function of ambient phosphorus levels are chlorophyll *a* concentration, algal biomass, water transparency and oxygen depletion in the hypolimnion.

The Microbiology Component specialists examined the effect of development on the occurrence and proliferation of a bacterium that causes *otitis externa*, the human disease known as "swimmer's ear". The prime indicator of swimming water quality in this study was the presence of the causal bacterium *Pseudomonas aeruginosa* in samples of lake water.

The Fisheries Component, which was divided into Net Productivity and Littoral Zone studies, was concerned with the impact of cottage development on the sport fish population in a lake. The Net Productivity scientists investigated the effect of angling pressure on the fishery and the relationship between angling effort, harvest and potential fish yield. An estimate of the maximum sustained yield for sport fish, based on lake morphometry and chemical factors, was used as the primary indicator of potential fish productivity in a lake. The cottager and non-cottager angling effort represented the pressure being placed on the fishery, whereas the fish harvest and biomass (stock) indicated the health of the sport fishery. Although data on all fish species were collected, the native lake trout *Salvelinus namayacush* was chosen as an indicator of the cold water fishery. This was done because they were found in many of the Study Area lakes, there was an angler preference for lake trout and they exhibited an extreme vulnerability to accelerated lake nutrient enrichment.

The Littoral Zone group investigated the effect of alterations in the nearshore habitat as a result of cottage development. The indicators were selected to give quantitative measures of the significance of habitat loss during key fish life-stages (Harker 1983). These indicators were: density of nests as a spawning indicator; density of young fish as a nursery indicator; and the densities of mature fish and prey organisms as indicators of the ability of the littoral zone to support fish in their feeding and resting areas.

The Wildlife Component scientists examined habitat modification associated with lakeshore cottages and the significance of these changes for representative animal species. Among the wildlife studied were songbirds, loons, hawks, small mammals, mink, deer, reptiles and amphibians. The species chosen occupy a wide range of ecological niches and therefore as a group are sensitive to the many direct and indirect impacts generated by development. The criteria for species selection in the Wildlife

Component were that they were (a) representative of major components of the wildlife community; and/or (b) valued highly by the public (for diverse reasons); and (c) present in sufficient numbers for study.

Several indices were used to gauge the extent of the impact on wildlife. The Area Development Index was used to predict the amount of vegetation disturbance and subsequent loss or gain of wildlife habitat (depending on the habitat preference of each species). Two indices of mink activity were also developed by the Wildlife Component. The Coefficient of Community was used as a measure of the extent of change in the native songbird species composition.

2.4 INTEGRATION PROCESS

The need to integrate the research findings into one predictive model which could be used to assist planners and resource managers was recognized from the outset of the Study. An exploratory literature review (Whitney et al 1978) was commissioned by the LCS Steering Committee to examine current approaches to environmental impact assessment modelling. From the group of relevant methodologies, the review identified those that were, in whole or in part, suitable to the objectives and design of the LCS and the integration process.

2.4.1 APPROACH TO INTEGRATION

The objective of the LCS was the development of a quantitative model, because of its usefulness in planning for lakeshore development. The research work had been done with this in mind and functional relationships had already been derived by the LCS components. Therefore qualitative models commonly used in environmental impact assessment were rejected.

The concept of a single carrying capacity for each lake was considered too rigid for the purposes of the study. This approach would preclude the consideration of potential variations in lake carrying capacity achieved through management strategies, such as modifications in subdivision design, restrictions on recreational activities (e.g. fishing) and reductions in nutrient loadings resulting from different types of sewage disposal.

A before-and-after approach was used by the LCS component researchers. However, for the purpose of integration, it was decided that temporal dynamics and interactions within the system over time should be explicitly included, to provide a more realistic simulation of actual conditions. For example, knowledge of the intermediate conditions on a lake would make it easier to explore the consequences of phased development. Also, if environmental impacts during the development period could be predicted at the outset, there would be sufficient lead-time to introduce suitable mitigative measures.

2.4.2 ADAPTIVE ENVIRONMENTAL ASSESSMENT AND MANAGEMENT

In their literature review, Whitney et al (1978) suggested that the most appropriate method of integrating the LCS component research was the approach (or at least some portion of it) developed by C.S. Holling and his colleagues at the Institute of Resource Ecology, University of British Columbia (Holling 1978). Whitney et al (1978) went on to describe several modifications to the approach which they considered more appropriate for the LCS. The concept of carrying capacity was recommended as the desirable framework for the integrated model.

After considering these proposals, the Integration Component researchers and the Study Steering Committee decided to adopt the Adaptive Environmental Assessment and Management (AEAM) modelling procedure to integrate the predictive methods developed by the components into a practical planning tool.

The AEAM approach prescribes the construction of a dynamic, computerized, simulation model, which explicitly accounts for the interactions (linkages) between compartments of the system under study. The complexity inherent in natural systems is minimized and the system is made more understandable by the use of simple yet robust mathematical relationships to describe the linkages in the model.

A simulation model is an abstraction of a real system, constructed to predict specific system responses. The projections indicate trends rather than exact solutions. This is useful in land use planning and resource management where an essential requirement of the model is to assist in environmental decision-making. The ease with which AEAM can be applied to the evaluation of resource management alternatives makes it helpful in resolving conflicts between opposing interests.

A critical ingredient of the AEAM approach is a series of intensive workshops during which a preliminary model is constructed and subsequently refined. The workshop participants (Appendix A) include scientists, planners, managers and policy advisors, from various disciplines (Figure 2.2). The composition of this group is designed to ensure that the model will reflect the needs and concerns of the potential users. The workshop dialogue encourages communication among those from different disciplines, which is valuable in defining their perspectives on lake planning.

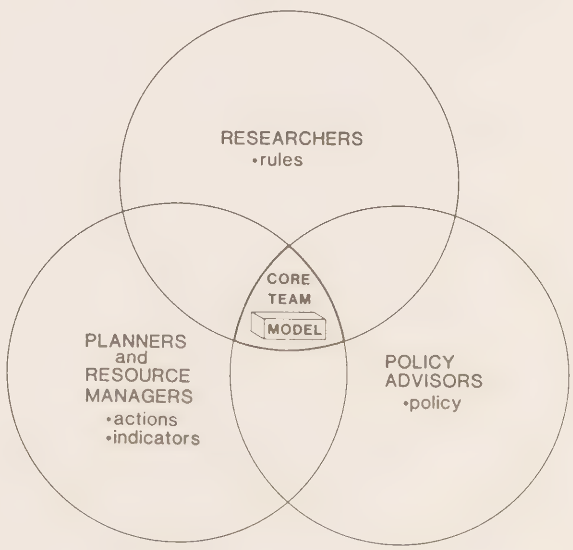


Figure 2.2 Workshop participants

3. ONTARIO LAKESHORE CAPACITY SIMULATION MODEL

3.1 MODEL STRUCTURE

The Ontario Lakeshore Capacity Simulation Model (OLCSM) is a type of simulation model sometimes referred to as a "state-transition discrete-time" model because the state of the system is described by variables that are changed at specified intervals of time. All natural events, such as precipitation, lake discharge and growth of the fish population, are implemented from the start of the simulation. The model contains a stochastic element in that random variation is applied to the annual nutrient loadings from precipitation and drainage, within the range observed in field studies.

Existing cottages are "built" at the start of the simulation. At any time thereafter, in response to a user-specified construction rate, cottages are built along the lakeshore on existing vacant lots and on lots in the proposed subdivision.

The parts of the OLCSM are illustrated in Figure 3.1. The inner workings of the model that provide the system dynamics, termed the "rules for change", describe in mathematical terms those processes and linkages between variables that are necessary for the prediction of impact on the environment. The rules are primarily a product of functional relationships derived by the LCS components from their research, supplemented with relevant data from the scientific literature and an intuitive understanding of the lake-watershed system. The levers represent the management strategies or "actions" that can be tested with the model. The dials depict the measures of the state of the system, or "indicators", which, when listed or plotted over time, comprise the model output.

The model is written in FORTRAN IV computer language and is run under the "operating system" called MTS (Michigan Terminal System), a general purpose time-sharing computer system. Access to the model is accomplished with a computer terminal link-up to the computer system.

The OLCSM is controlled by a computer program, Simulation Control System (SIMCON) which handles model input, output and record-keeping. SIMCON was designed for use with models that mimic changes over time.

The process of "running" the model involves a series of iterations (Figure 3.2). At each iteration the complete set of model equations is calculated, transforming the system state at time "t" to the system state at time "t + 1". The succeeding iterations are accomplished similarly. However, in each case, the system at time "t + 1" of the previous iteration becomes the new system at time "t".

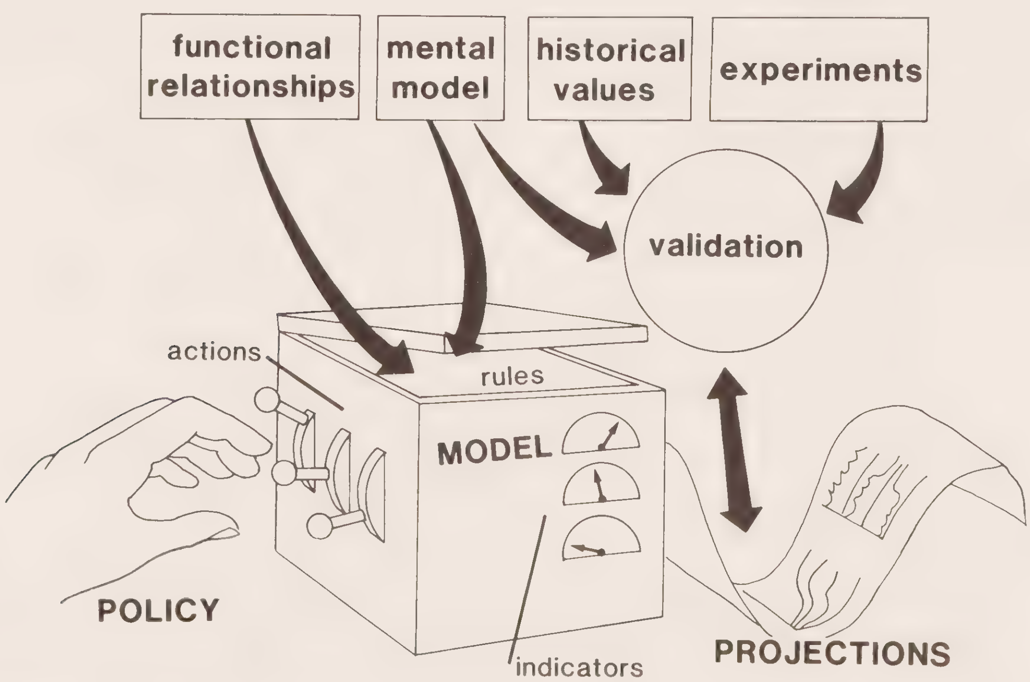


Figure 3.1 Modelling terminology

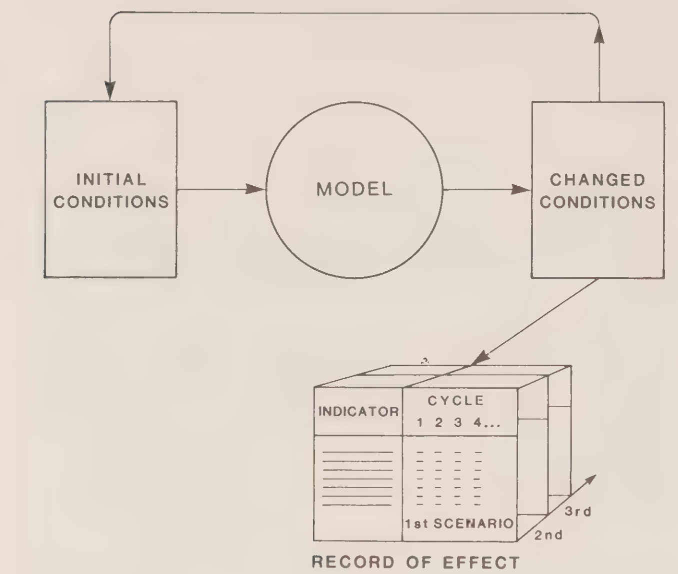


Figure 3.2 Model iteration and record-keeping for one cycle

SIMCON includes an intricate record-keeping system for retaining the values of each variable at each time-step, permitting the user to record changes in any variable during the time period of the simulation.

The model contains four subsystems, or submodels: Land Use, Water Quality, Fisheries and Wildlife-Habitat. Division of the system into subsystems facilitates examination of model coding and assumptions.

The structure of the submodels is prescribed by the format requirements of SIMCON. Each submodel is composed of four types of files: SOURCE, OBJECT, SET and COMMON (Figure 3.3). The user does not have direct access to the OBJECT file, as it is a machine-language translation of the SOURCE file. The SOURCE file contains the equations and other instructions which describe the system and indicate how the variables change over time. Variables and parameters are assigned values in the SET file. The COMMON file lists all the variables, along with the space in memory required for their storage. Initially, SET data are passed to the COMMON files. Each iteration uses these data, transforms values and retains the new and old values in the COMMON files.

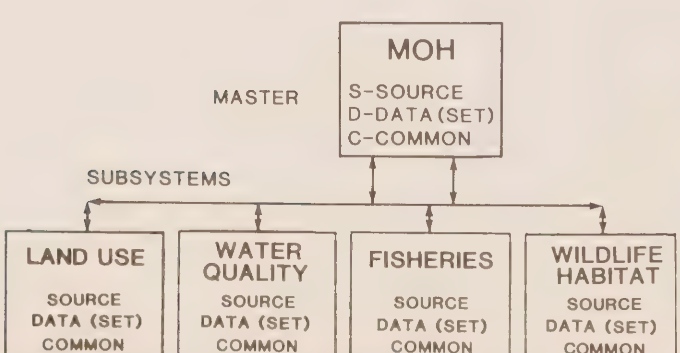


Figure 3.3 Ontario Lakeshore Capacity Simulation Model structure

A set of master files (MOH.S, MOH.D and MOH.C) links the submodels, allowing subsystem interactions and the sequencing of calculations between submodels. The master SET file contains, in addition to all the submodel SET files, the input data describing the specific lake characteristics that must be supplied to the model in order to evaluate conditions on a particular lake. The package of input data for each lake, describing the distribution of lots, forest cover, lake morphometry and other conditions specific to the lake, is termed a MACRO and is given a name, usually the name of the lake under study. By calling the MACRO, the user can obtain all the initial conditions for a specific lake.

The model output consists of numerical values for the environmental indicators (such as lake nutrient levels, fish stock or songbird habitat) that measure system behaviour. The values for these variables are calculated at each time-step for the duration of the simulation.

The model is designed as a practical tool with which to obtain, in the shortest possible time, the best information possible on the present and future state of the natural environment, as a basis for decision making. It provides an objective assessment of the changes that are likely to occur as a result of a set of conditions on a particular lake.

The responsible provincial and municipal agencies define the amount or percent of change in the lake environment that is acceptable. The priorities given to the indicators and the critical limits for the indicator values are drawn from these policies. A range of values may be used as guidelines, depending on resource plans and management policy.

3.1.1 SPATIAL AND TEMPORAL BOUNDARIES

3.1.1.1 SPACE

The submodels were developed principally from research in the Muskoka-Haliburton area of central Ontario. Therefore, the present version of the simulation model is valid only for this area.

The explicit spatial unit agreed on for the OLCSM is a single lake, surrounded by a 50-m band of shoreland. The band is broad enough to include single-tier lakeshore cottage development and the habitat for the selected wildlife-indicator species. To accommodate the songbird habitat, the lakeshore depth was extended to 100 m. Smaller spatial units were created for the Wildlife-Habitat Submodel by dividing the lakeshore and littoral zone into segments of uniform width for the collection of site-specific data on habitat types. These wildlife segments were also used to identify the location of cottage lots around the lake.

Nutrient loadings from upstream and the immediate watershed are explicitly included in the model. This is done to ensure that the level of development on upstream lakes, the types of land use within the watershed and the size of the watershed are considered in the calculation of the water quality indicator values (Figure 3.4). The model is structured so that chains of lakes can be evaluated. Beginning at the headwaters, the nutrient outflow from one lake is treated as nutrient input to the next lake downstream.

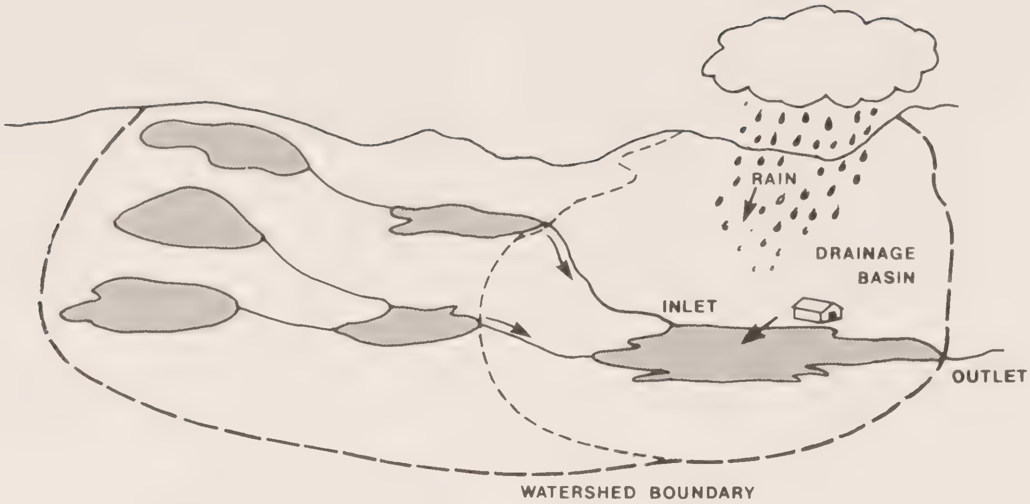


Figure 3.4 Spatial boundary: watershed including immediate drainage basin and upstream chains of lakes

3.1.1.2 TIME

The time step represents the increment of time between model calculations. In the present version, all submodels use an annual time-step, with the exception of the Water Quality Submodel (Figure 3.5). Within the annual time-step, monthly and seasonal events are calculated at the start of each year. In the Water Quality Submodel, the seasonal time-step is preferable because the phosphorus mass balance parameters can be more accurately estimated from seasonal data.

The length of each simulation (through time) is determined both by the needs of the user and the implications of the data. For predicting the impact of cottage development, a 20-year simulation is useful because it is an appropriate period for long-term planning. However, any time-period can be selected. To simulate the retention of sewage-derived phosphorus in the soil around the septic system, the length of time the soil retains its phosphorus-binding capacity must be carefully considered when setting a time horizon for the simulation. If the binding capacity of the soil continues beyond the simulation period, a reasonable prediction of the cottage impact on water quality would be impossible.

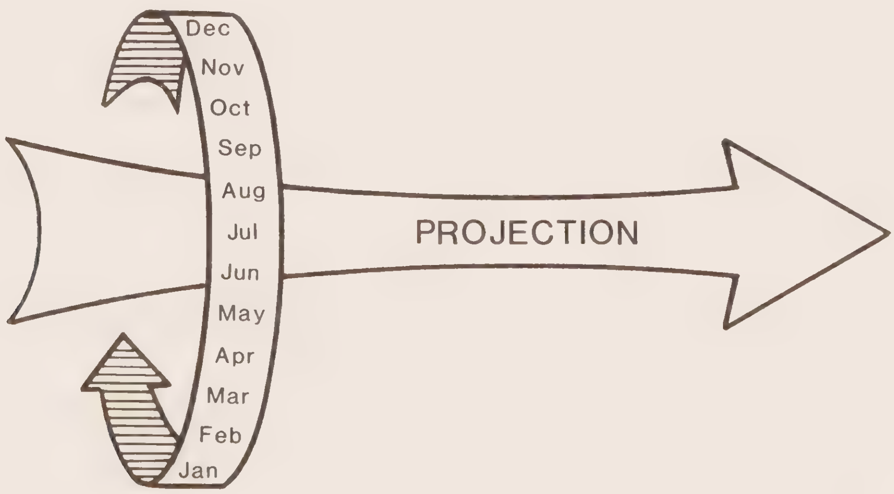


Figure 3.5 Time-step and length of simulation (e.g. 20-year projection)

3.1.2 INPUTS, ACTIONS AND INDICATORS

Some of the OLCSM variables remain relatively constant within the Study Area, e.g. annual phosphorus production per person. For these, uniform initial values are used for all lakes. However, lakes have many distinctive features for which specific values must be provided. The collection and organization of input data is described in a user's guide (Teleki and Herskowitz 1983a) which includes a data input form and instructions for entering the data into the computer. Most of the required input is available from established data bases, with the possible exception of information for the Wildlife-Habitat Submodel. If new wildlife-habitat data must be collected in the field, the survey could require several days, depending on the size of the lake.

The input data fall into four categories:

I. Required data: Values for some variables must be collected and entered into the computer in order to run the model (e.g. lake surface area, lake depth).

II. Useful data: Although the model is capable of calculating estimated values for certain variables, including actual values as input will improve the precision of model predictions (e.g. spring phosphorus concentration).

III. Optional data: Some parameters set in the model are based on average values in Muskoka-Haliburton (e.g. oxygen level at spring and fall turnover). Where possible, lake-specific data should replace the average values.

IV. Management data: The values of many variables may be altered in order to evaluate the effects of different management options (e.g. fishing season lengths).

System behaviour is measured in terms of meaningful and perceptible attributes termed indicators. These indicators comprise the model output and provide a measure of environmental impact. Those which are most useful to the user appear in the short list of the principal indicators in Table 3.1. The indicator values may be viewed in tabular form or plotted on graphs to illustrate the changes that occur in the environment over the period of the simulation.

Table 3.1 Principal indicators

Number of lakeshore cottages
Total number of annual user-days for lake
Summer dissolved oxygen minimum
Winter dissolved oxygen minimum
Seasonal and mean ice-free lake P concentration
Algal biomass over ice-free months (wet weight)
Zooplankton biomass over ice-free months (dry weight)
Proportion of water samples with PsA bacteria
Maximum Sustained Yield (MSY) for smallmouth bass and lake trout
Fish carrying capacity of lake (by species)
Fish biomass (by species)
Fishing effort (by species and season)
Fish harvest (by species and season)
Population indices for small mammals and songbirds
Hawk and loon nesting habitat loss
Mink activity index and disturbance factor

Selected management strategies, termed actions, may be introduced to mitigate the environmental impact of development. These actions pertain only to the parts of the ecosystem that can be managed and to modifications of the proposed development. When introduced or terminated at any time during the simulation, the actions correspond to the controls a manager might exercise as part of a management policy. Actions can be taken on any aspect of the lake-watershed system, e.g. modifications to the proposed plan of subdivision, the type of cottage access, the method of sewage disposal, or the length of the summer or winter fishing season (Table 3.2).

Table 3.2 Management strategies included in simulation model

SUBDIVISION RELATED
Regulate cottage density: lot width restrictions and number of lots allocated to each subdivision
Require staged development for proposed subdivisions
Control type of access, e.g. water, road, type of road and road maintenance
Control size of common beach
Negotiate cottage construction rate, e.g. through subdivider's agreement
Discourage cottage conversion from seasonal to permanent use
Modify subdivision design to avoid interference with critical wildlife habitat, e.g. location of road, lot or beach
WATER QUALITY RELATED
Limit phosphorus concentration in lake
Specify sewage system type and location for cottages
Require reduction of phosphates in sewage
FISHERIES RELATED
Shorten summer and/or winter angling season
Limit summer and/or winter catch
Encourage anglers to fish for underutilized species
Control public boat access
WILDLIFE-HABITAT RELATED
Reserve critical areas for deer, mink, loon, hawk and herptile habitat
Encourage cottagers to retain vegetation on lakeshore lots
Vary acceptable change in wildlife habitat

The model is designed to indicate the future repercussions of selected options which will assist the user in choosing the appropriate course of action. When several management scenarios have been simulated, the indicators from different scenarios can be compared. Ineffective actions can be identified and removed, leaving the most effective actions as a pool of viable management options.

Table 3.3 Data generated by each submodel

	Units
LAND USE	
total cottage-days for fishing seasons	
total user-days by month and by type of sewage system	
maximum number of swimmers at beach	
WATER QUALITY	
algal biomass	$mg \cdot L^{-1}$
chlorophyll *a* concentration	$\mu g \cdot L^{-1}$
Secchi depth	m
total phosphorus concentration (TP)	$\mu g \cdot L^{-1}$
areal hypolimnetic oxygen deficit (AHOD)	$mg \cdot m^{-2} \cdot day^{-1}$
zooplankton biomass	$mg \cdot m^{-3}$ (dry wt.)
proportion of water samples with *P. aeruginosa*	
minimum dissolved oxygen concentration	$mg \cdot L^{-1}$
Nitrogen:Phosphorus ratio	
FISHERIES	
lake trout biomass	$kg \cdot ha^{-1} \cdot year^{-1}$
smallmouth bass biomass	$kg \cdot ha^{-1} \cdot year^{-1}$
summer and winter catch	$kg \cdot ha^{-1} \cdot year^{-1}$
maximum sustained yield (MSY)	$kg \cdot ha^{-1} \cdot year^{-1}$
spawning habitat	m^2
fishing effort	angler-hrs $\cdot ha^{-1} \cdot yr^{-1}$
fish carrying capacity (K)	$kg \cdot ha^{-1}$
morphoedaphic index (MEI)	
WILDLIFE-HABITAT	
small mammal habitat	m^2
loon and hawk nesting sites	
littoral zone habitat for fish spawning	m^2
mink activity and disturbance index	
area development index (ADI)	
songbird habitat and breeding pairs	
critical areas flagged for preservation	
number of cottages in each year	

3.1.3 SUBMODELS AND THEIR LINKAGES

The research results of the LCS components are incorporated into four submodels, each designed to simulate a coherent part of the lake-watershed system (Table 3.3). Lake nutrient status and lake microbiology, which were investigated by separate components, are combined in the OLCSM Water Quality Submodel.

The linkages between submodels are summarized in the submodel interaction matrix (Table 3.4). For each submodel, it includes the information that is considered essential in describing the other model subsystems. For example, the Land Use Submodel generates cottage-day and user-day data for all other submodels because these variables are a measure of cottage development, the source of the impact. The data on oxygen and lake phosphorus concentrations generated by the Water Quality Submodel and passed to the Fisheries Submodel reflect the linkages between water quality and fish biomass. The Wildlife-Habitat Submodel calculates changes in the amount of fish spawning habitat which is subsequently used in the Fisheries Submodel to modify the fish carrying capacity of the lake. Also the number of cottages constructed in the Wildlife-Habitat Submodel is passed to the Land Use Submodel, for the predictions of total cottage use on the lakeshore.

The interactions between the submodels are selective rather than exhaustive. They are restricted to those that predict the behaviour of indicators related to the impact of lakeshore cottage development on the lake and surrounding shoreland.

The internal operation of each submodel and the linkages between submodels are described in the following four chapters.

Table 3.4 Submodel interaction matrix

FROM \ TO	LAND USE	WATER QUALITY	FISHERIES	WILDLIFE-HABITAT
LAND USE	all variables internal to submodel	user-days/month by sewage system type distances from beaches to nearest septic system maximum beach swimmers on day of highest use in each summer month	cottage-days/ month in summer and winter fishing seasons public boat access (non-cottager anglers)	total user-days in month of maximum use (July)
WATER QUALITY		all variables internal to submodel	Total Dissolved Solids (TDS) phosphorus concentration minimum oxygen concentration in hypolimnion zooplankton biomass	
FISHERIES			all variables internal to submodel	
WILDLIFE-HABITAT	number of cottages in each year (total for lake and by subdivision)		number of cottages/ 1000 m of shoreline	all variables internal to submodel

4. LAND USE SUBMODEL

Research by the Land Use Component planners and social scientists was concentrated largely on cottage use (Downing 1986). The methods developed to predict annual cottage use reflected differences between cottages with regard to type of access (e.g. road or water) and pattern of use (e.g. seasonal or year-round).

The Land Use Submodel applies these methods to estimate the amount of use a cottage will receive annually. The number of annual user-days is allocated to months and to the type of sewage system associated with the cottage. This information is used in the Water Quality Submodel to predict the amount of phosphorus generated from lakeshore cottages. Cottage use information is also passed to the other submodels.

The computer code which handles cottage construction is in the Wildlife-Habitat Submodel, because it is necessary to identify the location of cottages with respect to the wildlife segments for estimating impact on the wildlife community. However, it is more appropriate to describe cottage construction here, in association with cottage use.

4.1 COTTAGES

The model is programmed to handle five different subdivisions or sections of lakeshore, each with its own construction rate, start-up time and types of sewage system. The existing development consists of both cottages and vacant (undeveloped) lots, as each lot represents one potential cottage. The proposed subdivisions in the model represent cottage subdivision applications (real or hypothetical). Following the placement of existing cottages along the lakeshore at the start of the simulation, cottages are constructed annually on existing vacant lots and proposed lots.

The Wildlife-Habitat Submodel requires a fixed grid system in order to relate the location of cottages to the types of wildlife habitat around the lake. Therefore, the 50-m deep band of shoreland is divided into segments of uniform width which may be 50-200 m in width depending on the length of the lakeshore. Consistency is maintained between the location of cottages recorded in both the Wildlife-Habitat (Chapter 7) and Land Use Submodels, by simulating the development of wildlife segments rather than lots.

The cottage construction rate for each subdivision represents the proportion of undeveloped segments to be developed each year. For example, a subdivision may occupy 20 segments. If the construction rate is 0.5 and the subdivision begins in the first year, 10 segments would be developed in year one, 5 in year two, and so on, until the subdivision is completed. The segments are developed in a sequence assigned by the user. When a segment is developed, cottages are constructed on all lots located within the segment (Figure 4.1).

The "lot" referred to in this report is that part of the registered lot within the 50-m band of shoreland. Therefore, the lot size in the model is the width (m) of the registered lot multiplied by 50 m. For a given plan of subdivision, the user must calculate the average lot size in each wildlife segment. Lot size is important because it is interpreted in the Wildlife-Habitat Submodel as a factor affecting the extent of wildlife habitat disturbance.

In order to simulate cottage development and to predict cottage use, the following data are required:

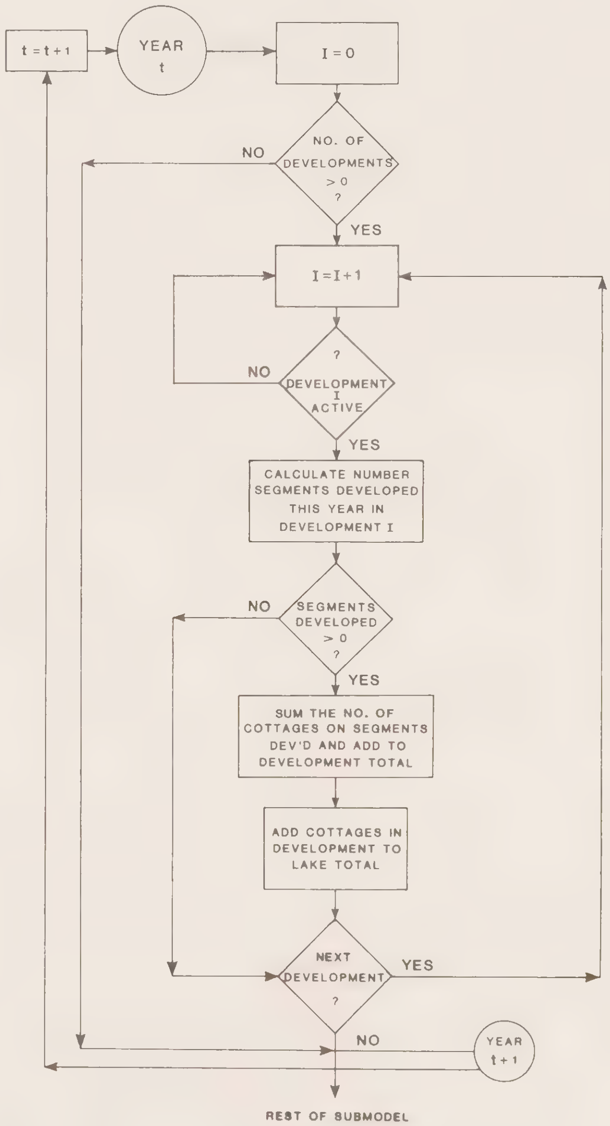


Figure 4.1 Flow chart of major events in the calculation of development ("development" refers to a subdivision or a section of lakeshore; I = development number)

A. To determine lot size and location of cottages:
   (a) number of existing and/or proposed lots in each wildlife segment,
   (b) average lot size for each wildlife segment with existing and/or proposed lots,
   (c) sequence of segment development,
   (d) rate of cottage construction, and
   (e) year construction is started in proposed subdivision.

B. To determine use of cottages:
   (a) type of access (road or water only),
   (b) summer and winter road access,
   (c) distance to nearest hard-surfaced road,
   (d) distance to nearest high order urban service centre, and
   (e) cottage typology (proportion of cottages in each of three cottage typology groups) from selected benchmark lake.

If special local conditions (Section 4.2.2) indicate that an increased (or decreased) rate of conversion from seasonal to permanent residences can be expected, the conversion rate is expressed as a decimal fraction plus 1 (e.g. 5% = 0.05 + 1 = 1.05).

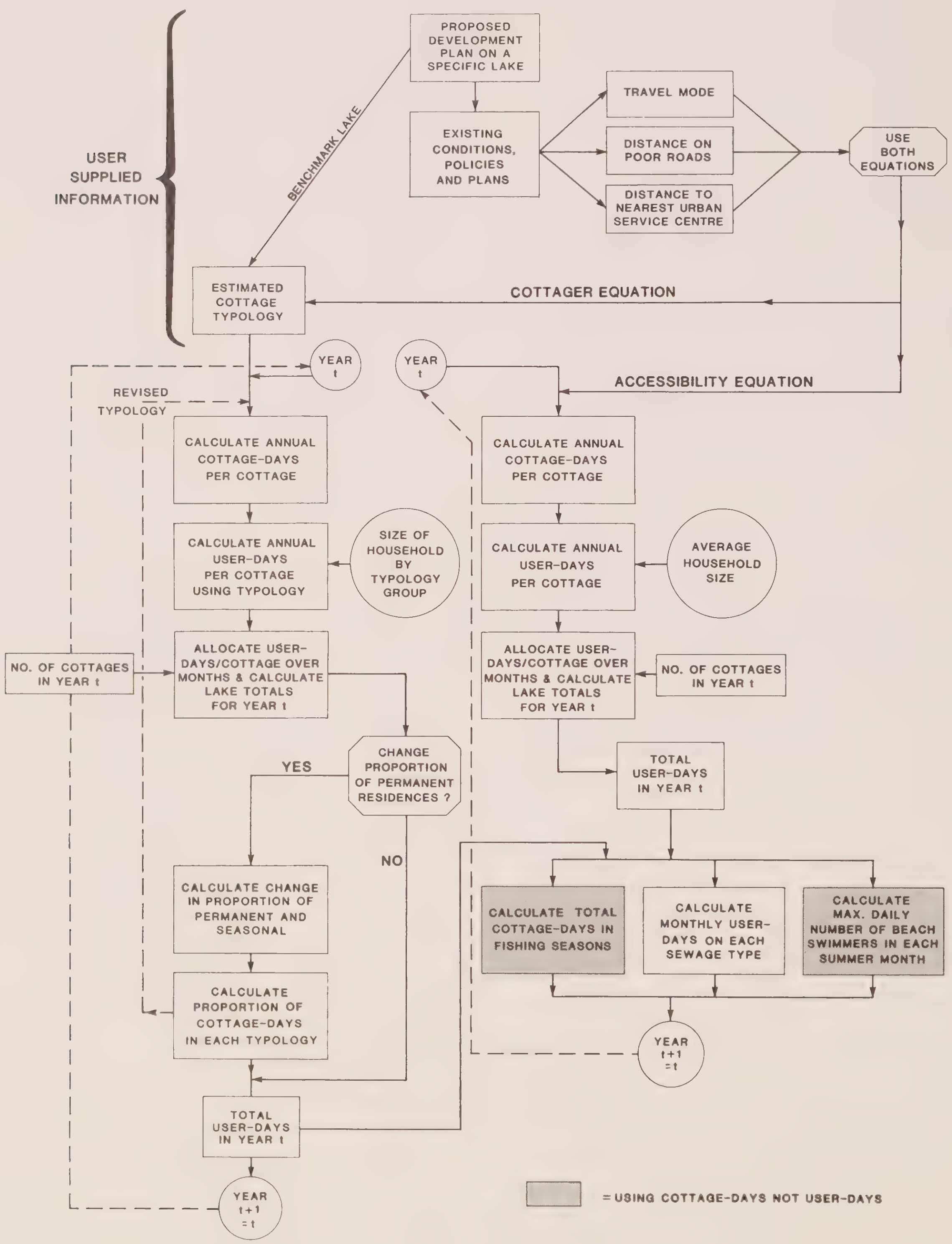


Figure 4.2 Land Use Submodel flow chart

4.2 COTTAGE-DAYS AND USER-DAYS

The major indicators of the Land Use Submodel that are employed in the other submodels are: 1) the cottage-day, one cottage in use for one day, and 2) the user-day, one cottage in use for one day by one person. These are expressed as the annual sum for all lakeshore cottages around a given lake.

The cottage use indicators are calculated from two equations, the Accessibility and the Cottager equations, developed by the Land Use Component (Downing 1986) to predict cottage-days. The subsequent conversion to user-days takes into account average household size and, in the case of the Cottager equation, also the cottage typology. A flow chart for the calculation of the cottage use indicators appears in Figure 4.2.

The accessibility variables can refer to a subdivision, a portion of a subdivision, a section of lakeshore or the lakeshore as a whole. The choice depends on the lake size, uniformity of travel mode, whether more than one "nearest urban centre" is involved and whether the distance on poor roads varies widely. For convenience, the descriptions of the accessibility variables in this chapter refer to a subdivision.

4.2.1 ACCESSIBILITY EQUATION

The Accessibility equation reflects variations in cottage use, as a function of selected factors related to cottage accessibility. The Accessibility equation below predicts annual cottage-days per cottage (COTDA):

$$\text{COTDA} = 83.105 + 93.452\,\text{TMODE1} + 22.655\,\text{TMODE2} + 21.972\,\text{TMODE3} - 0.529\,\text{DISTSUMH} + 0.322\,\text{MAXPOORX} \quad (4.1)$$

where TMODE1 = predominant travel mode to subdivision is car in summer and winter (0 = no, 1 = yes);

TMODE2 = predominant travel mode to subdivision is car in summer, other in winter (0 = no, 1 = yes);

TMODE3 = predominant travel mode to subdivision is boat in summer and other than car in winter (0 = no, 1 = yes);

DISTSUMH = mean distance from cottages in the subdivision to the nearest high order urban service centre (km); and

MAXPOORX = inverse of mean distance from cottages in the subdivision to the nearest paved road (km^{-1}).

Note: TMODE4 is the reference category in this regression equation. Only one of the TMODE variables can be equal to 1 for any given subdivision; the others equal 0.

The travel mode variable reflects higher use when the cottage is accessible by road and lower cottage use when it has water access only. Similarly, where the road is closed in winter, cottages are more difficult to reach and tend to be used less. The distance from the cottage to the nearest paved road also affects cottage use. The third variable relating to access is the distance to the nearest large urban service centre in the region. While short-stay cottagers, such as weekenders, have little or no need for local services, those who stay for longer periods need to replenish their supplies periodically. Year-round lakeshore residents need a wider range of services, including banks and medical facilities, and are more likely to locate closer to large urban centres in the region (Downing 1986).

To calculate the annual user-days for each cottage, the predicted number of cottage-days per cottage (Equation 4.1) is multiplied by the average household size of 3.067 people. Finally, the total number of user-days associated with the subdivision, in each simulated year, is the product of the annual user-days per cottage multiplied by the total number of cottages in the subdivision in that year. For the entire lakeshore, the totals for the various subdivisions are summed.

The Water Quality and Fisheries submodels require total user-days and cottage-days by month, respectively, for the entire lake. Using data from the Land Use Component, the average proportion of annual cottage use in each month is applied in the submodel to calculate the average monthly distribution of user-days and cottage-days.

4.2.2 COTTAGER EQUATION

The Cottager equation predicts cottage-days from the perspective of how the cottage is used. Three distinct patterns of cottage use were identified (Downing 1986) and the proportion of cottages in each of these three categories is termed the cottage typology. This can be thought of as a record of the end result of the diverse decisions made by cottagers concerning the use of their cottages. The decisions are based on a broad array of internal and external factors, such as socio-economic characteristics of the cottage household and cost of gasoline.

The three cottage typology groups are defined as follows:

Seasonal: Cottages used predominantly during the summer (July and August); on weekends in the spring, summer and fall; and on long weekends in the spring, summer and fall.

Extended Summer: Cottages with a high level of use (weekends and weekdays) throughout the spring, summer and fall.

Permanent Residents: Cottages used all year, as principal residences.

In order to predict cottage use for a proposed subdivision, the proportion of cottages in each typology group is estimated. This is accomplished by selecting an appropriate "benchmark" lake from the pool of 36 benchmark lakes, for which the cottage typology proportions are known. The benchmark lake's cottage typology is then applied to the total cottage-days for the proposed subdivision. The method of selecting the appropriate benchmark is based on matching the accessibility characteristics of the target lake or subdivision with those of a benchmark lake, assuming that differences in accessibility are associated with differences in cottage use. The benchmarking

process has been documented in detail in the Land Use Component Technical Report (Carroll et al 1983).

The Cottager equation below predicts annual cottage-days per cottage (COTDA):

$$\text{COTDA} = 68.312 + 278.578\ \text{PERMANENT} + 102.042\ \text{ESUMMER} + 21.291\ \text{TMODE1} + 12.623\ \text{TMODE2} - 0.191\ \text{DISTSUMH} \quad (4.2)$$

where PERMANENT = proportion of cottages in typology group Permanent Residents (as determined from benchmark lake);

ESUMMER = proportion of cottages in typology group Extended Summer (as determined from benchmark lake);

TMODE1 = predominant travel mode to subdivision is car in summer and winter (0 or 1);

TMODE2 = predominant travel mode to subdivision is car in summer, other in winter (0 or 1); and

DISTSUMH = mean distance from the cottages in the subdivision to the nearest high order urban service centre (km).

Note: The SEASONAL typology group is the reference category in this regression equation and is included as the constant 68.312.

The conversion of total annual cottage-days for the lake-shore to user-days, using the Cottager equation, proceeds as follows:

$$\text{total annual user-days} = \sum_{i=1}^{3} TCD_t * P_t(i) * \text{HSIZE}(i) \quad (4.3)$$

where Σ = summation

$*$ = multiplication symbol

i = 1 – Seasonal
2 – Extended Summer
3 – Permanent Residents

TCD_t = total annual cottage-days associated with the cottages in the subdivision in year t, i.e. predicted cottage-days per cottage (Equation 4.2) * number of cottages;

$P_t(i)$ = proportion of total cottage-days associated with cottage typology i in year t (Table 4.1); and

HSIZE(i) = average household size for typology i (Table 4.1).

The Water Quality Submodel requires user-days on a monthly basis. Since cottages within each typology group have different monthly use patterns, the allocation of total user-days to each month, accomplished as follows, will reflect these variations in cottage use.

$$\text{total monthly user-days (m)} = \sum_{i=1}^{3} TCD_t * P_t(i) * \text{HSIZE}(i) * \text{FRAC}(i,m) \quad (4.4)$$

where FRAC(i,m) = proportion of total annual cottage use (user-days) allocated to month m for typology i (Figure 4.2).

TCD_t, $P_t(i)$, HSIZE(i) are defined in Equation 4.3

As the land use equations were developed from survey data, the increased cottage use associated with conversion from seasonal to permanent residences and from permanent to seasonal are included in the predictions of cottage use. Therefore, the historical rates of cottage conversion are implicit in the Land Use Submodel. However, if local conditions suggest that an unusually high (or low) conversion rate is likely to occur, such as might result from new employment opportunities, an appropriate conversion rate can be implemented in the model.

The user would specify an annual rate of transition from typology groups 1 and 2 (spring, summer and autumn users) to typology group 3 (permanent residents), i.e. the rate of cottage conversion. If the proportion of permanent users exceeds 0.6, the annual rate of transition is set to zero, i.e. cottages used all year will not exceed 60% of the total. The consequence of a change in the typology proportion is a disproportionate change in the total cottage-days associated with each typology. This occurs because cottages in the permanent residents group are occupied for more days during the year than those in the other typology groups (Table 4.1).

Table 4.1 Average number of days a cottage is occupied per year (CDAY) and average household size (HSIZE) for each cottage typology group (i)

Cottage Typology Group (i)	CDAY(i)	HSIZE(i)
1 = Seasonal	71.0	3.55
2 = Extended Summer	175.2	2.63
3 = Permanent Residents	363.1	2.55

4.3 TIME FRAME FOR PREDICTIONS

When the model is applied, it is important that the predictions remain valid for as long as possible — ideally, for the life of the cottages. Realistically, this is not attainable but it is possible to take policies and plans for the future into account in assigning values to the travel mode variables and in selecting an appropriate benchmark lake. For example, if the proposed subdivision has no access road at present but an approved transportation plan for the area provides for a road, then the travel mode would be recorded as "car" for all or part of the year. If, on the other hand, the transportation plan shows no road, then the travel mode would be recorded as "boat". Similarly, where an official plan indicates that permanent use may be prohibited and the zoning by-law specifies a "limited services" district for the area in which the proposed subdivision is located, it must be assumed that the road would be closed in winter. This would be taken into account in the choice of a benchmark lake.

The land use equations rely on existing cottage use patterns, which were established over time. The Land Use Component recommends periodic monitoring to provide a factual basis for judging whether the extent or rate of change merits a new survey.

4.4 ALLOCATION OF USER-DAYS TO TYPES OF SEWAGE DISPOSAL

Associated with each simulated cottage development on the lake is a specification of the mode of sewage disposal. This information is available from the subdivision proposal. Provisions are made for three kinds of disposal systems: lagoon, septic system, and holding tank. Pit privies are included with septic systems as both release sewage effluent into the soil. For the Water Quality Submodel, the predicted number of user-days must be allocated to each sewage disposal type, on a monthly basis, to provide data for estimating the annual amount of cottage-generated phosphorus. The allocation is calculated as follows:

$$\text{monthly sewage user-days}(ij) = \sum_{k=1}^{NDEV} TMUD_t(i) * PCOTT_t(k) * STYPE(k,j) \quad (4.5)$$

where NDEV = number of subdivisions;
$TMUD_t(i)$ = total user-days in month (i), in year (t) (from Equation 4.4);
$PCOTT_t(k)$ = number of cottages in subdivision (k), in year (t); and
STYPE(k,j) = proportion of cottages in subdivision (k) using sewage disposal type (j) where 1 = lagoon, 2 = septic system, 3 = holding tank.

4.5 COTTAGE-DAYS FOR FISHING SEASONS

The Fisheries Submodel requires the total number of cottage-days for the summer (May – September) and winter (January – March) fishing seasons as part of the calculation of fishing effort. This is computed simply as the sum of the monthly information over the relevant periods.

4.6 NUMBER OF SWIMMERS

The number of swimmers at public or common beaches is required by the Water Quality Submodel, in order to predict the proportion of beach water samples with *Pseudomonas aeruginosa* (PsA), the pathogen that causes human ear infections. In the current submodel, two approaches are provided for estimating the number of swimmers in each month:

1) The first approach is to estimate swimmer density along the entire lakeshore, based on the total number of cottages on the lakeshore. The following equation is applied in this case to predict the maximum number of swimmers (MAXSWIM):

$$MAXSWIM = NO.\ COTT * POCC(i) * PSWIM2(i) * UCOTT(i) * USWIM(i) \quad (4.6)$$

where NO. COTT = total number of lakeshore cottages;
POCC(i) = proportion of cottages occupied on day of highest cottage use in summer month i;
PSWIM2(i) = proportion of cottages with swimmers on day of highest cottage use in month i;
UCOTT(i) = average household size for cottages occupied on day of highest cottage use in month i; and
USWIM(i) = proportion of swimmers (from cottages) actually swimming between 11:00 a.m. and 2:00 p.m.

2) The second approach focusses on common beaches, each identified with the subdivision they serve. The number of swimmers is based on the proportion of cottages in the subdivision with swimmers who use the common beach. The following modification of Equation 4.6 is used to predict the maximum number of swimmers at a common beach in subdivision kx (MAXSWIMB):

$$MAXSWIMB = COTI(kx) * POCC(i) * PSWIM(i) * UCOTT(i) * USWIM(i) \quad (4.7)$$

where kx = subdivision number;
COTI(kx) = number of cottages in subdivision kx;
i = months 6-9 (June – September); and
PSWIM(i) = proportion of cottages with swimmers at a public or common beach on day of highest cottage use in month (i).

POCC, UCOTT and USWIM are defined in Equation 4.6.

The values used in the equations and the sources of these data are listed in Table 4.2. The occupancy data (POCC) and household size (UCOTT) vary by month. The last term, proportion of swimmers (from cottages) actually swimming between 11 a.m. and 2 p.m. (USWIM) incorporates two factors: distribution of swimmers throughout the day, and people in the water versus people on the beach. Additional terms could be used to handle an estimate of the proportion of swimmers who put their ears under water and the ratio of cottager swimmers to non-cottager swimmers for the lake.

Table 4.2 Parameter estimates used to calculate the number of swimmers at public or common beaches and in the water along the shore, for the selected day of highest cottage use in each summer month.

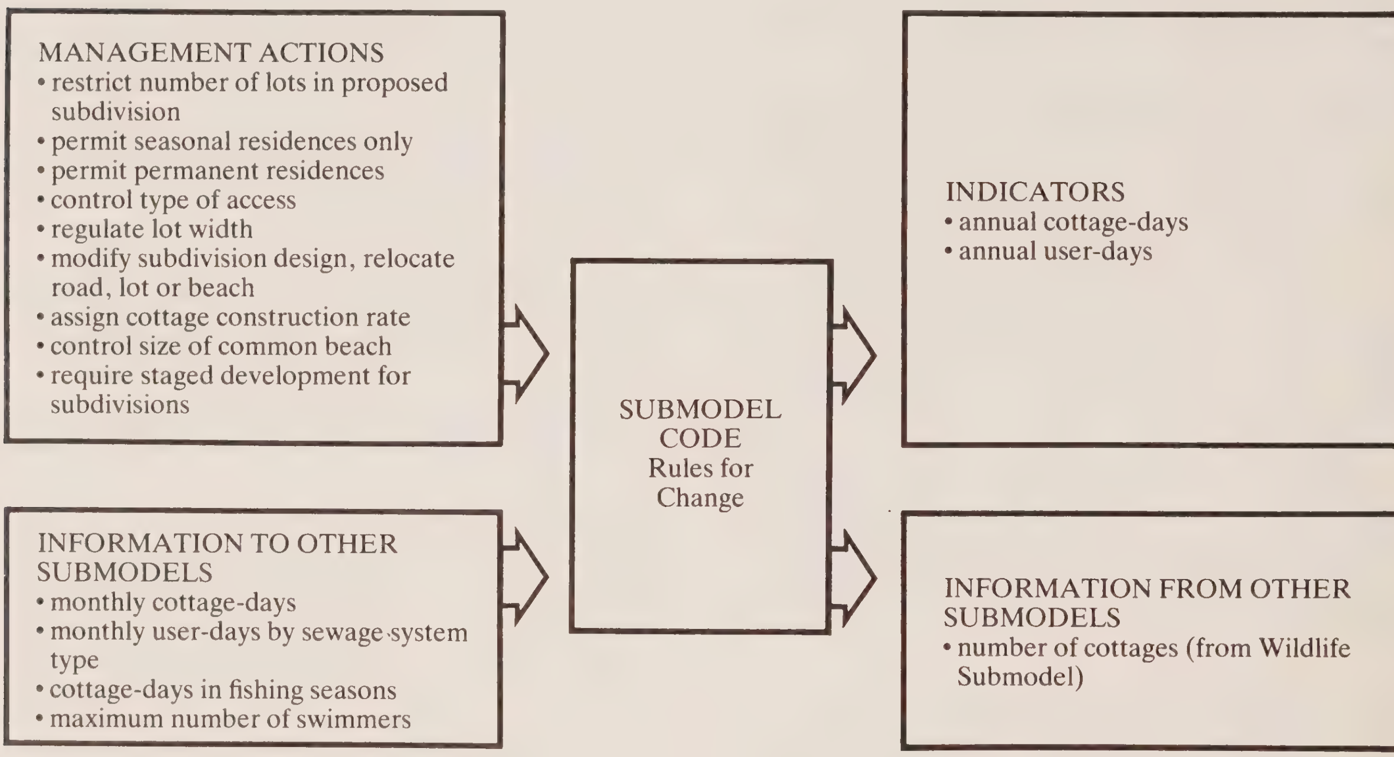
MONTH[1]	POCC[2]	PSWIM[3]	PSWIM2[4]	UCOT T[5]	USWIM[6]
June	0.66	0.32	0.96	2.1	0.5
July	0.87	0.32	0.96	3.6	0.5
August	0.89	0.32	0.96	3.6	0.5
September	0.84	0.32	0.96	3.3	0.5

[1] Day of highest use in the month is associated with the last two weekends in June, Dominion Day in July, the Civic Holiday in August and Labor Day in September.
[2-5] Land Use Component estimate, based on 1978 Survey of Lakeshore Residents.
[6] Microbiology Component estimate, based on knowledge of study lakes.

Information on maximum swimmer density is used in the Water Quality Submodel to calculate the proportion of water samples with PsA bacteria, for the months June to September. The frequency of occurrence of the PsA bacteria in the water, in the area used for swimming, affects the amount of swimmer contact with these organisms and the risk of contracting the ear infection *otitis externa*. If a common beach is associated with subdivision kx, the model estimates the frequency of occurrence of the bacteria at a specific beach. If there is no beach associated with a particular subdivision, the indicator becomes the lake shoreline background PsA value.

4.7 SUMMARY

The Land Use Submodel provides a measure of cottage use, the major source of environmental impact from lakeshore cottage development. The submodel can simulate conditions on a lake with existing and proposed cottages and can alter these conditions in accordance with user-specified management actions. The Land Use Submodel generates the key land use indicators as well as specific information required by other submodels to determine the impact on the environment. The following input-output box diagram summarizes these submodel elements.



5. WATER QUALITY SUBMODEL

The Water Quality Submodel is divided into two parts: one designed to predict the nutrient level, or trophic status, of the lake and the other concerned with the health-related microbiology of lake water.

The focus of the trophic status portion of the submodel is phosphorus, as it is the nutrient most frequently controlling algae and plant production in north temperate lakes. High phosphorus levels are associated with the pollution of lakes and the decrease in water quality, i.e. water clarity and dissolved oxygen. The microbiology part of the submodel predicts the presence in lake water of the pathogen *Pseudomonas aeruginosa*. It is the causal organism of the human ear disease, *otitis externa* and is common enough in recreational lakes to be a public health concern. The major events in the Water Quality Submodel appear in the flow chart (Figure 5.1).

5.1 SUBMODEL TIME-STEP

The Water Quality Submodel is capable of making seasonal or annual predictions. A seasonal time-step is preferable, as this permits examination of the interactions of different cottage typologies and/or sewage treatment systems (e.g. lagoons with release in the spring) with seasonal patterns of flushing and natural phosphorus loading.

Indicators dependent on phosphorus concentrations, i.e. algal biomass, chlorophyll *a*, Secchi depth, and zooplankton biomass, are average seasonal values since there is insufficient understanding to justify monthly calculations.

The four seasons (relative to deep lake water conditions) used in the submodel are:

(a) Spring (March 16 – May 15)
(b) Summer (May 16 – October 31) stratified lake
(c) Fall (November 1 – November 30)
(d) Winter (December 1 – March 15) non-stratified lake

The lengths and starting times of these seasons vary somewhat with the size of the lake and other physical parameters. At present, the submodel uses the average dates listed above, but it has been structured to permit the user to specify the duration of each season.

5.2 TOTAL PHOSPHORUS

Lake phosphorus concentration is the net result of natural and artificial inputs, retention in the lake sediments and release through outflow streams (Figure 5.2). Total phosphorus predictions are based on the assumption that the lake is a homogeneous water mass. In fact, thermal stratification produces concentration gradients and mixing delays.

From a knowledge of the geology and land use in the drainage basin, an estimate is made of the total phosphorus exported per unit area which, combined with the drainage area, provides an estimate of the total phosphorus supplied to the lake from the drainage basin. To calculate the total natural phosphorus supply to the lake, the drainage basin phosphorus is added to the phosphorus in precipitation falling directly on the lake, and the phosphorus contribution from inflow streams.

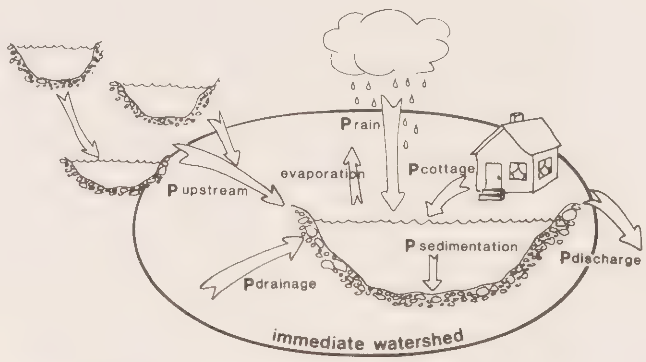


Figure 5.2 Input and output of phosphorus

The artificial (cottage-generated) phosphorus supply is calculated from the yearly per capita estimate of phosphorus in household sewage (0.81 kg/yr) and the predicted number of user-days for cottages on a given lake. The proportion of the cottage-generated phosphorus that will reach the lake is related to the number of user-days associated with each type of sewage disposal system. Holding tanks, for example, release no phosphorus to the lake because the tanks are pumped out periodically and the sludge is removed to treatment plants outside the watershed. Sewage lagoons, which are not commonly used for cottage development, are settling basins that are partially drained in the spring, at which time about 15% of the sewage-phosphorus is released. Septic and aerobic systems and pit privies, the dominant sewage disposal systems, release effluent to the soil and subsurface water. The amount of phosphorus reaching the lake is calculated from the phosphorus binding capacity of the soil reservoir and the phosphorus leakage rate.

Changes in the lake total phosphorus concentration are calculated according to the following equation, adapted from Dillon and Rigler (1975):

$$\Delta [P] = \frac{(P_{RAIN} + P_D + P_{UP} + P_{CL})}{V} - \delta [P] - \rho [P] \quad (5.1)$$

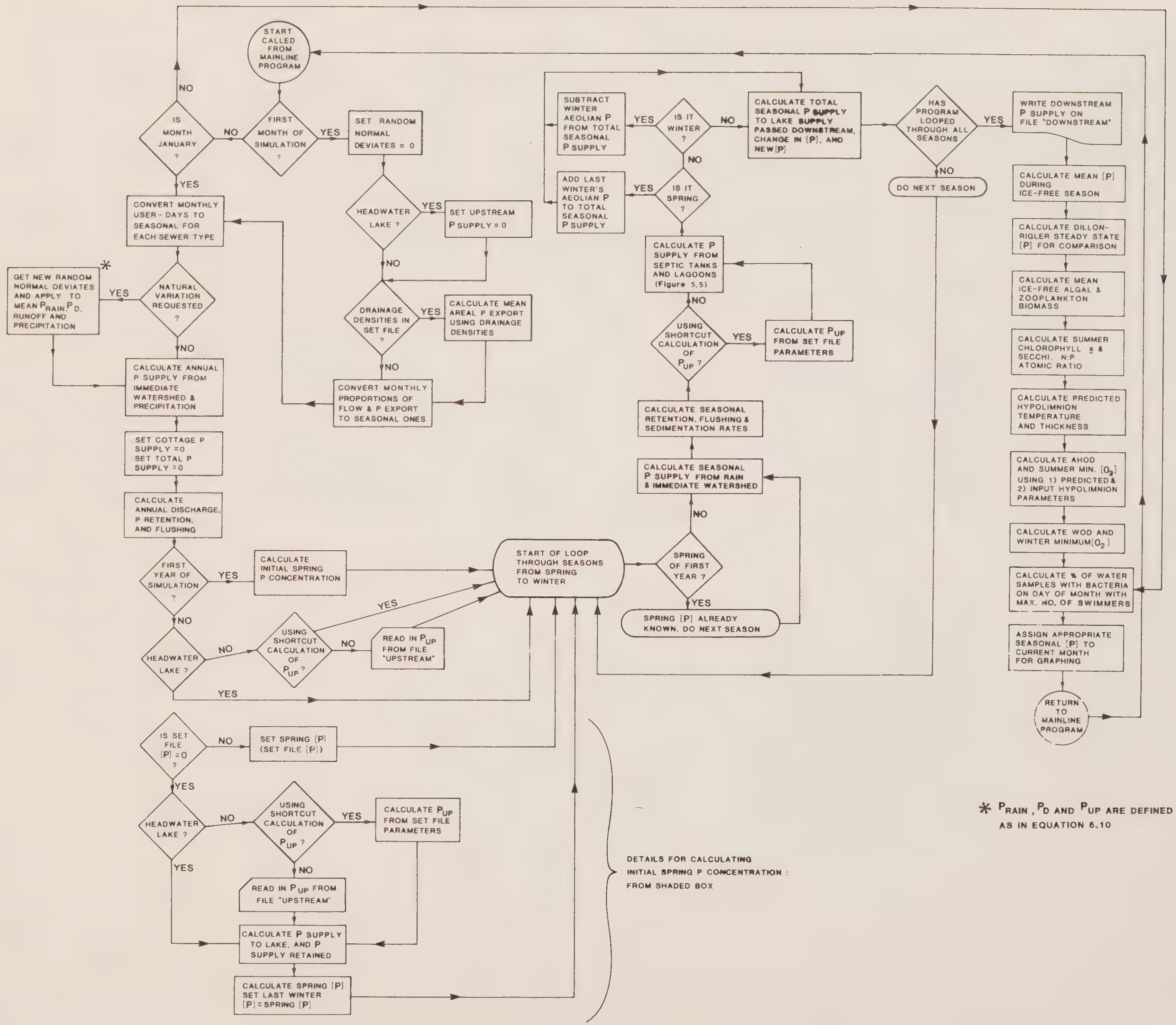


Figure 5.1 Water Quality Submodel flow chart

where $\Delta[P]$ = change in total phosphorus concentration ($mg \cdot m^{-3}$) over a given season;
P_{RAIN} = supply of total phosphorus in precipitation during the given season (mg);
P_D = natural supply of total phosphorus to the lake from its immediate drainage basin, during the given season (mg);
P_{UP} = supply of total phosphorus from lakes upstream during the given season (mg);
P_{CL} = supply of total phosphorus from cottages that reaches lake during the given season (mg);
V = volume of water in lake (m^3);
$[P]$ = previous season's total phosphorus concentration ($mg \cdot m^{-3}$);
δ = phosphorus sedimentation rate ($season^{-1}$); and
ρ = lake flushing rate ($season^{-1}$).

5.3 PRECIPITATION P LOAD

The annual precipitation (aeolian) supply of P is calculated by randomly choosing a number from a normal distribution with a mean of 37.3 $mg \cdot m^{-2} \cdot yr^{-1}$ and a standard deviation compiled from four years of aeolian P data (Scheider, pers. comm. 1982). If no annual variation is desired (useful for first-cut model runs), the standard deviation is set to zero, as in the example in Appendix C. When natural variation in the model is switched on, precipitation, runoff, nutrient loads from precipitation and watershed export of nutrients all become stochastic variables, i.e. varying in a quasi-random fashion to mimic natural occurrence.

The monthly proportions of the annual phosphorus supply, based on data (1976-80) collected by the Trophic Status Component (Figure 5.3), are converted to seasonal proportions. The precipitation phosphorus supply to the lake during the winter is accumulated as phosphorus stored in snow on the ice cover. It is then added, as snow melt, to the aeolian phosphorus falling on the lake in the spring.

5.4 IMMEDIATE WATERSHED P LOAD

At present the submodel calculates the export of P from the immediate watershed by one of two methods, depending upon whether information is available on the watershed sub-basin drainage. Where possible, the P export from each sub-basin is calculated by the regression equation[1] of Kirchner (1975):

$$\text{Phosphorus export } (mg \cdot m^{-2} \cdot yr^{-1}) = 1.32 + 5.54 * \frac{\text{length of all streams (km)}}{\text{sub-basin area (km}^2)} \quad (5.2)$$

The total P supply from the specified sub-basins is calculated by multiplying the P export for each sub-basin by the area of the sub-basin and summing over all sub-basins. For the sections of the watershed that are not drained by streams, the average of the sub-basin P export values is applied to the surface area. The total P supplied from drainage is the sum of the supplies from all parts of the immediate drainage basin. The above method assumes that the watershed is forested and underlain by igneous rock.

The annual drainage basin P supply is prorated over the months and seasons according to Figure 5.4 which is based on data (1976-80) provided by Scheider (pers. comm. 1982).

The alternative method is to apply an average P export value for areas underlain by igneous rock. Depending on whether the watershed is forested or consists of forest plus pasture (i.e. cleared on more than 15% of the watershed), the P export values of 4.7 $mg \cdot m^{-2} \cdot yr^{-1}$ and 10.2 $mg \cdot m^{-2} \cdot yr^{-1}$, respectively, are used in the model (Dillon and Rigler 1975).

[1] This equation will be replaced by an equation recently developed by the Trophic Status Component (Dillon, pers. comm. 1982) which was not available in time for inclusion in this version of the OLCSM. The simplified version of the new equation predicts the annual gross phosphorus export from the watershed as a function of peat (% area), grade (% area), bedrock (% area) and outwash plain (% area).

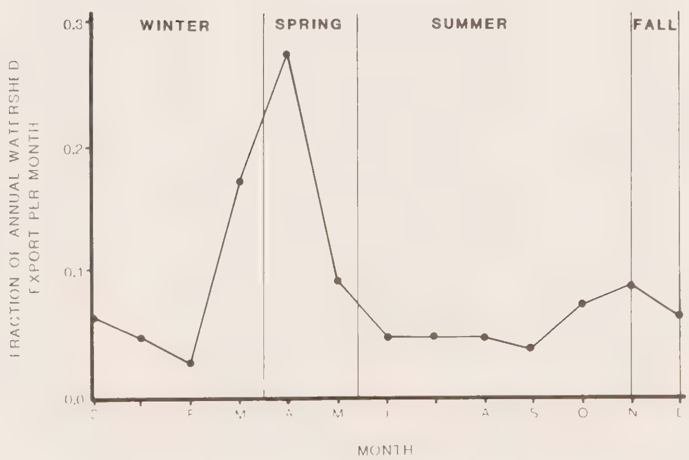


Figure 5.3 Mean monthly distribution of bulk precipitation phosphorus supply over 4-year period (1976-80)

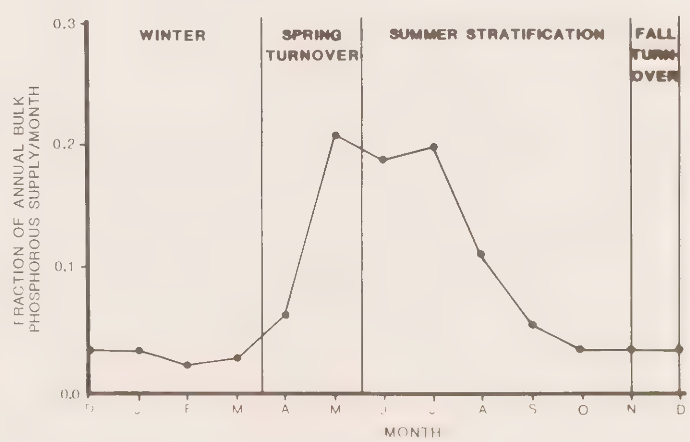


Figure 5.4 Mean monthly distribution of watershed phosphorus export over 4-year period (1976-80)

5.5 UPSTREAM P LOAD

Where development is proposed for a watershed containing more than one lake, phosphorus balance calculations proceed from the highest (headwater) drainage basin down to the lowest. Predictions of trophic status indicators are made for each lake in the chain. From the initial conditions and proposed developments on a headwater lake, a time series of phosphorus supply released downstream is calculated and stored. These data then provide the upstream P supply for the simulation of conditions (over the same time period) in the next lake downstream. The time series of phosphorus releases from the second lake serves as the upstream P supply for the third lake, and so on.

One advantage of this approach is the ability to examine downstream impacts of particular policies. For example, development on a well-flushed headwater lake may not significantly increase phosphorus concentrations there, but could have a decided impact on a larger, poorly drained, downstream lake. The model is capable of simulating any configuration of connecting lakes.

Calculation of the upstream P supply by the above method is only feasible when information on lake and watershed morphometry, as well as present and future land use, is available for each lake in the chain. Since such information is often not available, a "short-cut" approach can be used to estimate upstream contributions of phosphorus. Upstream supplies are calculated from two user-specified parameters: the mean concentration of phosphorus in the next lake upstream, and the proportion of watershed runoff that drains through these upstream lakes. The seasonal P supply from upstream (P_{UP}) is then:

$$P_{UP} = YQ * FUP * FQ * [P]_{UP} \quad (\mathrm{mg \cdot season^{-1}}) \tag{5.3}$$

where YQ = annual discharge through lake ($\mathrm{m^3 \cdot yr^{-1}}$);
FUP = fraction of watershed drainage through upstream lakes;
FQ = fraction of YQ occurring in a given season; and
$[P]_{UP}$ = mean phosphorus concentration in upstream lakes ($\mathrm{mg \cdot m^{-3}}$).

5.6 COTTAGE P LOAD

At present, calculations of the cottage phosphorus supply to the lake are performed in two steps:

(a) calculation of the amount of phosphorus generated by the lakeshore cottages, depending on the types of sewage disposal systems (Table 5.1);
(b) calculation of the amount of phosphorus actually reaching the lake each season.

The total P generated by cottages each season (PCOTT) is calculated by:

$$PCOTT_i = 2191.8\ \mathrm{mg \cdot user^{-1} \cdot day^{-1}} * \quad (\mathrm{mg \cdot season^{-1}})$$

$$\sum_{i=1}^{12} USERM_i * (1\text{-}SEWRET_i) \tag{5.4}$$

where $USERM_i$ = user-days on sewage disposal method i in given season; and
$SEWRET_i$ = fraction of cottage-generated P removed (removal function) from watershed under method i.

The smaller the SEWRET value, the more P release is assumed. For example, for a lagoon system the SEWRET value is 0.85. Therefore 15% of the sewage phosphorus would be released. With respect to lagoons, the user can specify the season(s) during which the lagoon is to be emptied.

Table 5.1 Sewage disposal system categories

Type of system	Fraction of P supply removed from watershed (SEWRET)	Time of release to lake
1. Lagoon	0.85	any season (user defined)
2. Septic tank, aerobic system or pit privy	0.0	continuous
3. Holding tank	1.0	no release

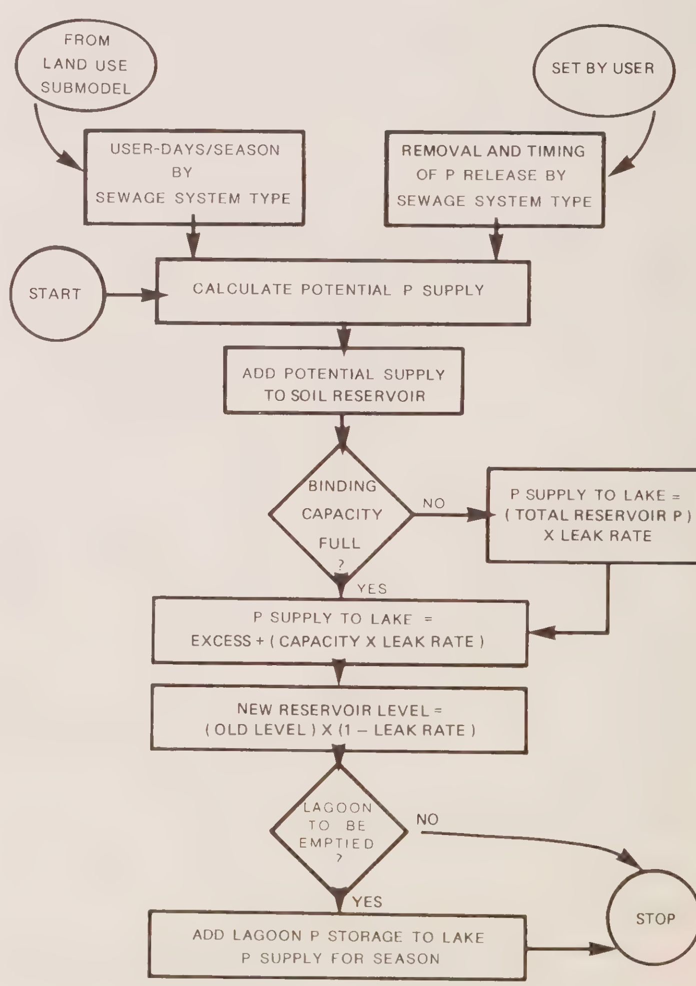


Figure 5.5 Algorithm for calculating seasonal phosphorus supply from cottages to lake

To explore the consequences of different soil types, the submodel allows the specification of a phosphorus binding capacity in the soil, expressed in kg of phosphorus binding capacity per cottage, and a transport or leakage rate from soil storage (RLEAK), expressed as a fraction of the capacity for each season. A maximum phosphorus binding capacity per cottage (2.4 kg phosphorus) equivalent to three years use by permanent residents was arbitrarily assigned as an initial condition. However, by setting RLEAK = 1, it was also assumed (for the base runs) that all the phosphorus within the soil storage would be transported to the lake within one year. Without modifying any of the assumptions, the base runs of the model represent a 'worst case' from the point of view of cottage phosphorus supply. The phosphorus actually transported to the lake in a given season is calculated according to the algorithm shown in Figure 5.5.

5.7 WATERSHED HYDROLOGY AND PHOSPHORUS RETENTION

The concentration of nutrients in lake water is the net result of the mass of material deposited in the lake, minus the material removed from the water column through flushing and sedimentation.

The annual outlet discharge YQ, in $m^3 \cdot yr^{-1}$, is calculated using the formula developed by Dillon and Rigler (1975):

$$YQ = [A_d * r + A_o (P_r - E_v)] * [10^4\ m^2 \cdot ha^{-1}] \qquad (5.5)$$

where A_d = total drainage basin area including all upstream drainage (ha);
A_o = lake surface area (ha);
r = mean annual areal runoff ($m \cdot yr^{-1}$) (assumed value 0.4);
P_r = amount of annual precipitation (m) (assumed value 0.9); and
E_v = amount of annual evaporation (m) (assumed value 0.6).

If the total watershed area is greater than 10 times the lake surface area, the net increase in the lake water due to precipitation falling directly on the lake, $A_o(P_r - E_v)$, is negligible and is therefore omitted from the equation.

The phosphorus supply from drainage is calculated only for the lake's immediate watershed because P drainage through upstream lakes is included separately. The water loading for the lake is based on the entire drainage basin upstream of the lake. In Lake Boshkung, for example, the entire drainage basin is 63 times the area of the lake's immediate watershed. This contrasts with headwater lakes, such as Harp or Blue Chalk, where the immediate watershed is equal to the entire drainage basin.

When natural variation is invoked in the simulation, runoff and precipitation vary in the same direction as aeolian and watershed phosphorus supply. Evaporation remains constant since no information is available to indicate to what degree variation in evaporation is correlated with rainfall. The mean values for the variables listed above were obtained from Lakeshore Capacity Study data and from the literature (Dillon and Rigler 1975, Scheider 1978).

Annual discharge is converted to seasonal discharge from the monthly distribution (Figure 5.6) which was extracted from data provided by Scheider (1976-1980). The seasonal discharge divided by the lake volume yields the flushing rate, ρ, which is the number of times the lake's volume is completely replaced in that season. The ρ rate is an important determinant of the seasonal change in phosphorus concentration, as shown by Equation 5.1.

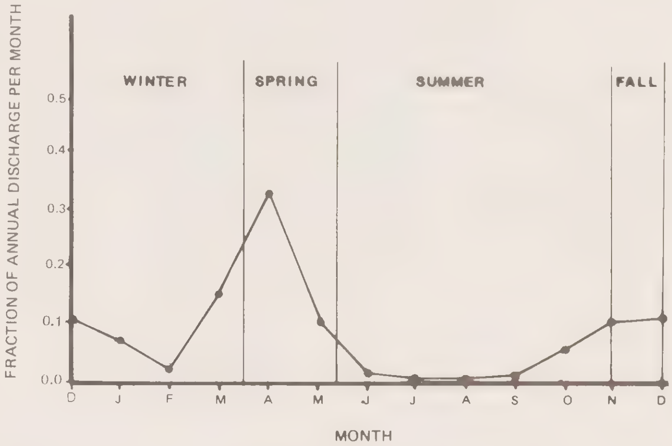


Figure 5.6 Mean monthly distribution of annual discharge over 4-year period (1976-80)

The other important parameter in Equation 5.1 is the phosphorus sedimentation rate, δ. Although δ is difficult to measure directly, it is related by Equation 5.6 to the empirically predictable retention coefficient, R, the fraction of the phosphorus supply not lost via outflow:

$$\delta = \frac{\rho R}{(1\text{-}R)} \quad \text{(Dillon and Rigler 1974)} \qquad (5.6)$$

Equation 5.6 is valid for any time-step, provided that R and ρ have been calculated for a similar time period. Prior to the development of the OLCSM, an annual time-step was used. Annual retention coefficients (YR) have been estimated using various empirical equations (Ostrofsky 1978), the simplest of which is:

$$YR = \frac{V}{V + YQS} \quad \text{(Dillon and Kirchner 1975)} \qquad (5.7)$$

where YR = annual phosphorus retention coefficient (rate) (proportion $\cdot yr^{-1}$);
V = annual phosphorus settling velocity ($m \cdot yr^{-1}$);
YQS = annual areal water loading = YQ/AREA ($m \cdot yr^{-1}$);
YQ = lake outlet discharge ($m^3 \cdot yr^{-1}$); and
AREA = lake surface area (m^2).

The original formulation[2] of Equation 5.7 had a constant value for V of 13.2. However, retaining this constant in the calculations produced highly inaccurate results. Therefore, an interim equation was developed which brought the model indicators more in line with the actual measurements used as a comparison. This formulation

[2] Dillon (pers. comm. 1982) has developed a better formula for the calculation of phosphorus retention. The new equation predicts phosphorus retention in a lake as a function of the lake flushing rate. This equation was not available in time for incorporation in this version of the OLCSM.

was based on a stepwise forward inclusion regression procedure (SAS 1979) with data from eight Trophic Status Component study lakes which have no lakeshore development. The following equation was derived and is presently used in the model:

$$V = 0.0074\ ALD + 1.927 \qquad (5.8)$$

where V = annual phosphorus settling velocity ($m \cdot yr^{-1}$);
ALD = [YPLOAD/AREAL * 10,000] * DEPTH;
YPLOAD = total phosphorus supply to lake ($mg \cdot yr^{-1}$);
AREAL = lake surface area (ha); and
DEPTH = mean lake depth (m).

The seasonal retention coefficient (R) was estimated by the equation:

$$R = \frac{RFIT}{RFIT + QS} \qquad (5.9)$$

where QS = seasonal areal water loading = Q/AREA;
Q = seasonal discharge = YQ * FQ (Equation 5.4);
AREA = lake surface area (m^2); and
RFIT = net rate of sedimentation of total phosphorus (in metres per season). This may be interpreted as a sinking rate through the water column.

Since no empirical relationship existed for seasonal retention coefficients, the RFIT values were fitted by trial and error matching of the steady state concentrations predicted by the seasonal time-step submodel and predicted by the equation:

$$P^* = \frac{J(1\text{-}YR)}{YQ} \quad \text{(Schneider 1978)} \qquad (5.10)$$

where P^* = steady state concentration ($mg \cdot m^{-3}$); and
J = supply to lake ($mg \cdot yr^{-1}$) summed over 4 seasons. This value includes ($P_{RAIN} + P_D + P_{UP} + P_{CL}$)
where P_{RAIN} = aeolian phosphorus;
P_D = phosphorus from drainage;
P_{UP} = phosphorus from upstream; and
P_{CL} = phosphorus from cottages.
YR and YQ are defined in Equation 5.7.

Based on an understanding of the relative rates of sedimentation, and the parameter fitting, the following proportions are applied in apportioning the annual settling velocity to the four seasons.

Season	Proportion of annual sedimentation RFIT	Settling Velocity V assuming an annual V of 13.2 $m \cdot yr^{-1}$
Spring	0.00	0.00
Summer	0.64	8.44
Fall	0.04	0.53
Winter	0.32	4.23

If Equation 5.9 is substituted in Equation 5.6, the sedimentation rate for a season is equal to RFIT for that season divided by the mean depth of the lake. Not surprisingly, deep lakes lose phosphorus to the bottom sediments at a slower rate than shallow lakes.

5.8 ALGAL BIOMASS AND CHLOROPHYLL *a*

A weighted average of the total phosphorus concentrations in spring, summer and fall is used to predict mean algal biomass during summer stratification. The submodel uses a relationship (Figure 5.7) based on measurements in the Kawartha Lakes – Bay of Quinte region, reported by Nicholls and Dillon (1978), as no such relationship has been developed for Ontario lakes in the Precambrian Shield.

The mean summer chlorophyll *a* concentration is estimated using the Dillon-Rigler (1975) formulation (Figure 5.8). Their equation uses the spring phosphorus concentration in the formula:

$$\log CHLOR = 1.45 \log SCPLK(1) - 1.14 \qquad (5.11)$$

where CHLOR = chlorophyll *a* concentration ($\mu g \cdot L^{-1}$); and
SCPLK(1) = spring phosphorous concentration ($\mu g \cdot L^{-1}$).

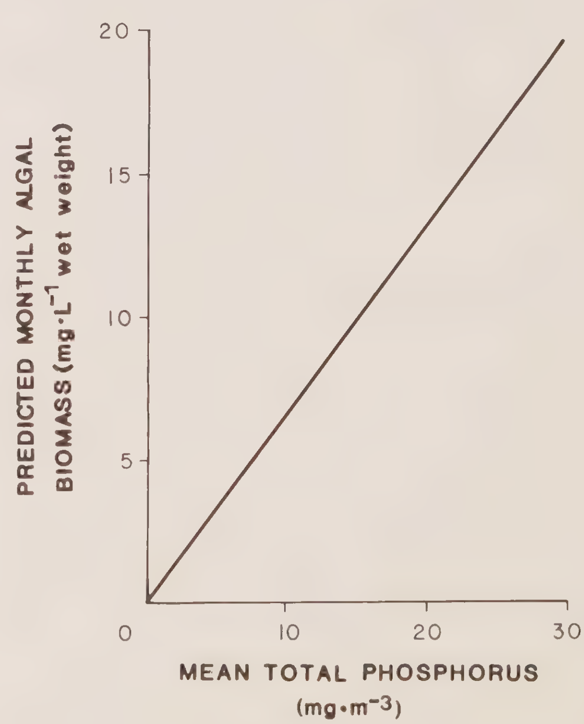


Figure 5.7 Mean algal biomass and mean total phosphorus concentration during ice-free months (from Nicolls and Dillon 1978)

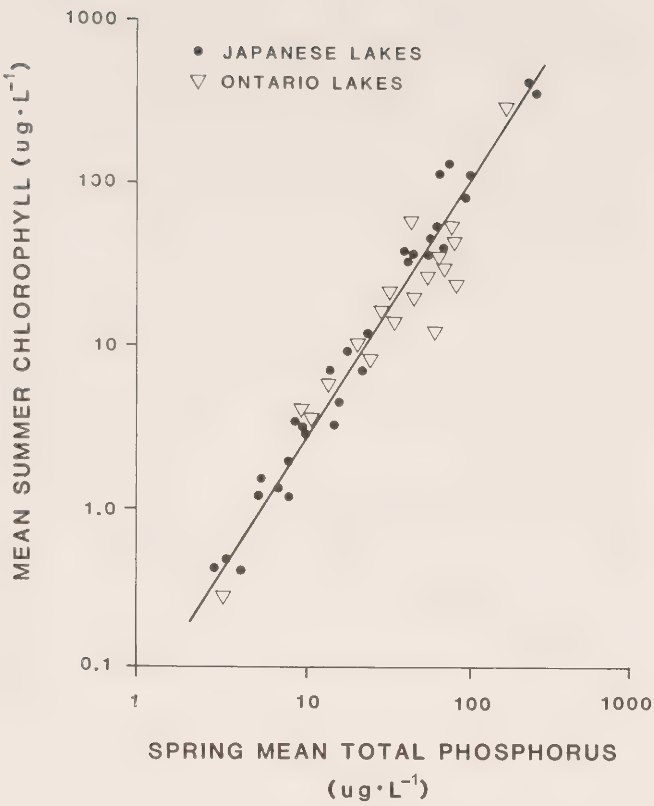


Figure 5.8 Relationship between mean spring total phosphorus and mean summer chlorophyll *a* (from Lorenzen 1978)

5.9 SECCHI DEPTH

Transparency, as measured by Secchi disc depth, is presently estimated from the chlorophyll *a* concentration (Scheider, pers. comm.) by:

$$\text{SECCHI} = \frac{5.21}{\text{CHLOR}^{0.41}} \tag{5.12}$$

where SECCHI = mean summer Secchi depth (m); and
CHLOR = chlorophyll *a* concentration ($\mu g \cdot L^{-1}$).

5.10 EDIBLE ALGAL BIOMASS

As a lake becomes eutrophic, a decreasing proportion of the algal biomass is consumed by zooplankton. This is because the ratio between P and nitrogen is disturbed enough to cause growth of inedible algal species. Therefore, the food base for planktivorous fish cannot be assumed to continue to increase linearly with increasing nutrients. A nitrogen balance equation following the form of Equation 5.1 (neglecting algal fixation of atmospheric nitrogen) is used to predict approximate changes in nitrogen concentration:

$$\Delta[N] = \frac{(N_{RAIN} + N_D + N_{UP} + N_{CL})}{V} - \delta[N] - \rho[N] \tag{5.13}$$

where $\Delta[N]$ = change in total N concentration ($\mu g \cdot L^{-1}$) over a given season;
N_{RAIN} = supply of total nitrogen in precipitation during the given season (mg);
N_D = natural supply of nitrogen from immediate drainage basin during the given season (mg);
N_{UP} = supply of nitrogen from lakes upstream during the given season (mg);
N_{CL} = supply of nitrogen from cottages reaching the lake during the given season ($\mu g \cdot L^{-1}$);
δ and ρ = were assumed to be the same for N as for P;
[N] = previous season's nitrogen concentration ($\mu g \cdot L^{-1}$); and
V = volume of lake (m^3).

The development of blue-green algal blooms (mostly inedible) is assumed to be dependent upon (a) lake phosphorus concentrations of at least 10 $mg \cdot m^{-3}$, and (b) N:P atomic ratios[3] less than 14. Figure 5.9 illustrates these hypotheses, with the probability of blooms varying directly with phosphorus concentration and inversely with the N:P ratio (Scheider, pers. comm. 1980). Phosphorus concentrations between 10 and 30 are given lines of intermediate slope.

Total nitrogen concentration is set initially to 664 $mg \cdot m^{-3}$, which together with the initial total phosphorus concentration of 15 $mg \cdot m^{-3}$, yields a starting N:P atomic ratio of 20:1.

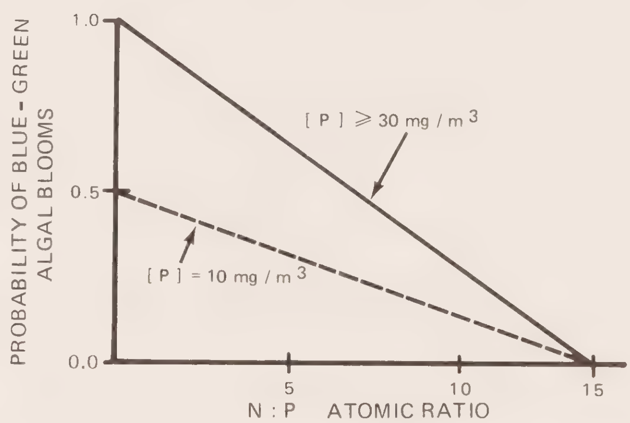


Figure 5.9 Probability of blue-green algal blooms for given N:P ratios

5.11 ZOOPLANKTON

The mean density of zooplankton during the summer lake stratification period is predicted from the mean ice-free phosphorus concentration [P] (a weighted average of spring, summer and fall concentrations) by the regression equation:

$$\text{mean zooplankton biomass } (mg \cdot m^{-3} \text{ dry wt.}) = 4.5 * [P] + 38 \tag{5.14}$$

This equation is based on Ministry of the Environment data obtained from lakes in the Kawartha Lakes – Bay of Quinte region. Since the carrying capacity for trout is presently predicted directly from lake nutrient concentration, bypassing the zooplankton level, this prediction is calculated but not used in the Fisheries Submodel. However, it remains in the submodel because of its potential future use.

[3] atomic ratio = 0.45 × weight ratio

5.12 OXYGEN

The areal hypolimnetic oxygen deficit (AHOD) is calculated each year using the multiple regression equation developed by Cornett and Rigler (1979):

$$AHOD = -277 + 0.5\ R_p + 5.0\ T_H^{1.74} + 150\ \ln(Z_H) \qquad (5.15)$$

where AHOD = areal hypolimnetic oxygen deficit ($mg\ O_2 \cdot m^{-2} \cdot day^{-1}$);
R_p = areal phosphorus retention ($mg \cdot m^{-2} \cdot yr^{-1}$) = [P] * YQ * YR/(AREA * (1-YR));
[P] = mean ice-free phosphorus concentration ($mg \cdot m^{-3}$);
YQ = annual outflow discharge ($m^3 \cdot yr^{-1}$);
YR = annual phosphorus retention coefficient (range 0-1);
AREA = area of lake (m^2);
T_H = mean temperature of the hypolimnion (°C);
ln = natural logarithm; and
Z_H = mean thickness of the hypolimnion (m).

A second equation developed by Charlton (1980) is also programmed into the model. It differs from Equation 5.15 in that chlorophyll *a* is used instead of phosphorus retention.

$$AHOD = 3.80 * [FCHLOR * (Z_H/(50. + Z_H)) * 2((T_H - 4.0)/10)] + .12 \qquad (5.16)$$

where FCHLOR = $1.15\ (CHLOR^{1.33})/(9.0 + 1.15 * CHLOR^{1.33})$
= chlorophyll *a* factor derived from the Equation 5.11.
Z_H and T_H are defined in Equation 5.15.

A switch is provided so that the user can work with either equation, permitting a comparison of the results of the two methods (Appendix B).

5.12.1 HYPOLIMNION TEMPERATURE AND THICKNESS

Two approaches are used in estimating the mean temperature and thickness of the hypolimnion.

The direct approach requires T_H and Z_H as input data and assumes that they remain constant over time. This method can be used on any lake for which temperature profile data over the summer stratification season are available.

The indirect method uses a hypothetical set of curves to predict mean hypolimnetic temperature and thickness (Figures 5.10 and 5.11, respectively). The mean temperature of the hypolimnion is assumed to be inversely proportional to the maximum depth of the lake, and directly proportional to the Secchi depth. This is because greater transparency results in increased heating of the hypolimnion. These relationships are represented by the equation:

$$T_H = 4.0 + (50.0 - Z_{MAX}) * 0.04 * SECCHI \qquad (5.17)$$

where T_H = mean temperature of the hypolimnion (°C)
Z_{MAX} = maximum depth (m); and
SECCHI = mean summer Secchi depth (m).

The hypolimnetic temperature is plotted (Figure 5.10) against Secchi depth for various maximum depths; a maximum temperature of 25°C is assumed. The effect of this relationship on the AHOD calculation is that nutrient enrichment will cause lowered transparencies. This, in turn, will lower hypolimnetic temperatures and thereby reduce the oxygen deficit (theoretically via reduced bacterial activity).

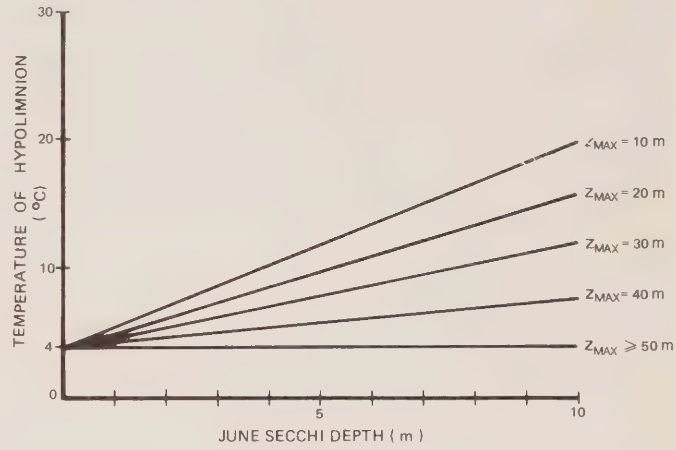


Figure 5.10 Temperature of hypolimnion as a function of Secchi depth and maximum depth (Z_{MAX})

The mean thickness of the hypolimnion (Z_H) is assumed to operate essentially in reverse to the hypolimnetic temperature, that is, the thickness increases with maximum depth and is inversely proportional to Secchi depth at transparencies less than 5 m (Figure 5.11). This relationship tends to counteract the hypolimnetic temperature function discussed above. As transparency drops, the thickness of the hypolimnion increases, resulting in a greater AHOD.

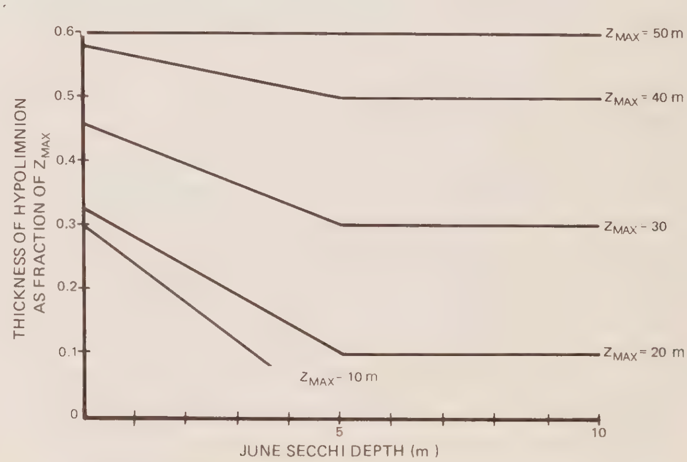


Figure 5.11 Mean thickness of hypolimnion as a function of Secchi depth and maximum depth (Z_{MAX})

A minimum hypolimnetic thickness of 1 m is assumed, to avoid negative values in the ln (Z_H) term of Equation 5.15. This assumption only becomes relevant for highly transparent, shallow lakes where Z_{MAX} is less than 10 m and SECCHI is greater than 5 m.

The net effect of the relationships presented in Figures 5.10 and 5.11 on AHOD is shown in Figure 5.12, for lakes of various depths. At maximum depths, the net contribution of the hypolimnion temperature and thickness terms of the AHOD decreases as transparency decreases.

Hypolimnetic oxygen depletion generally is accelerated by nutrient enrichment. For this acceleration to take place (Figure 5.12) in the submodel, all of the increase in AHOD must originate from the phosphorus retention term. AHOD predictions, using both methods, are compared in Appendix B.

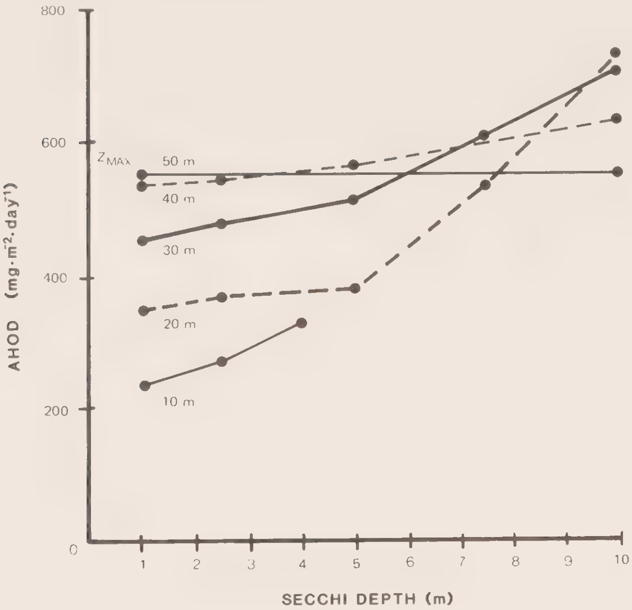


Figure 5.12 Net effect of changed transparency on the AHOD

Dissolved oxygen concentrations at spring and fall turnover are set by the user and typically vary from 10 to 12 $mg \cdot L^{-1}$. The two estimates of AHOD (direct and indirect) are converted from oxygen as $mg \cdot m^{-2} \cdot day^{-1}$ to $mg \cdot L^{-1} \cdot month^{-1}$, and multiplied by the length of the summer stratified season (generally 5.5 months) to yield the expected decreases in oxygen concentration over the summer period. Subtracting these monthly expected decreases from the input spring overturn concentration yields the summer minimum hypolimnetic oxygen concentration.

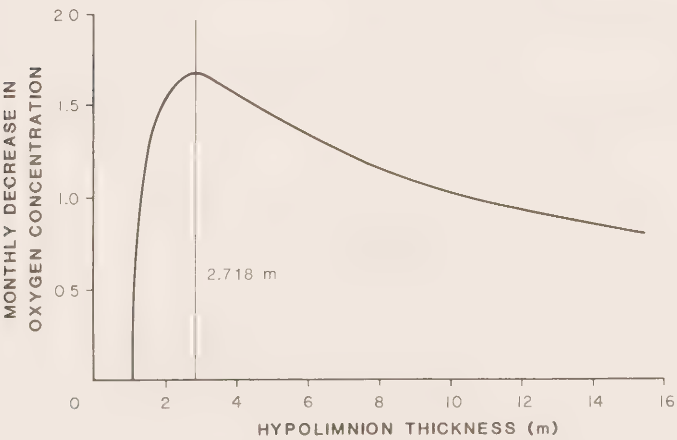


Figure 5.13 Effect of hypolimnion thickness on monthly decrease in oxygen concentration (independent of hypolimnetic temperature and phosphorus retention)

Since the AHOD calculated from Equation 5.15 is divided by the hypolimnion thickness (Z_H) to determine the volumetric oxygen depletion rate, the oxygen decrease is proportional to $\ln Z_H / Z_H$. The consequences of this proportionality are shown in Figure 5.13.

The maximum monthly decrease in oxygen concentration (other factors being equal) occurs at a hypolimnion thickness of 2.718 m. Where the hypolimnion thickness is less than 2.718 m, the predicted summer oxygen concentration drops off sharply. This reduction is a consequence of the regression Equation 5.15.

The minimum winter oxygen concentration is calculated from the winter oxygen deficit (WOD), which is estimated by the multiple regression of Welch et al (1976):

$$WOD = 315.7 + 20.8(\overline{Z}) - 38.2(SECCHI) - 1.5(\rho) - 1.8(\overline{Z}_{MAX}) \qquad (5.18)$$

where
- WOD = winter oxygen deficit ($mg \cdot m^{-2} \cdot day^{-1}$ for entire lake);
- $\overline{Z}$ = mean depth of lake (m);
- $SECCHI$ = mean summer Secchi depth (m)[4];
- ρ = flushing rate (yr^{-1}); and
- Z_{MAX} = maximum depth of lake (m).

The areal units of WOD apply to the whole lake, whereas those for AHOD apply only to the hypolimnion. As there is no appreciable stratification, the winter dissolved oxygen concentrations apply to the whole lake volume.

5.13 BACTERIA

The proportion of water samples in a public swimming area containing detectable levels of *Pseudomonas aeruginosa* which causes *otitis externa*, an infection of the human ear, was found (Burger 1983) to be affected by:

(a) the distance to the nearest septic tank, drainage ditch, lagoon, or other source of the bacteria,
(b) the density of swimmers using the beach, and
(c) the temperature of the water.

[4] Welch et al (1976) do not specify which season's Secchi depths should be used in the equation.

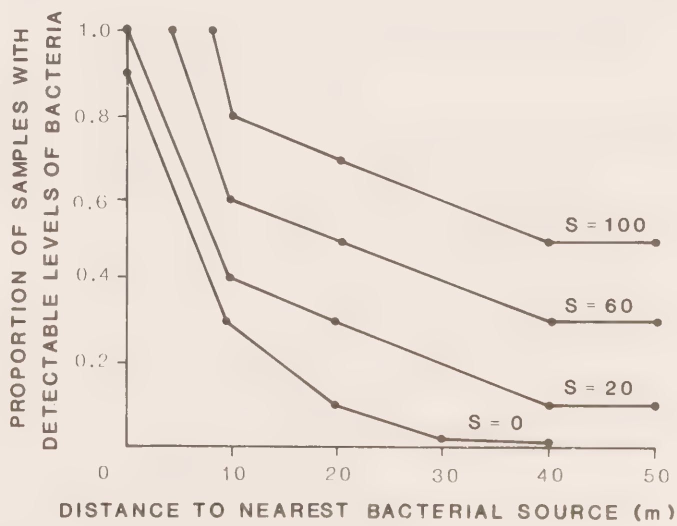


Figure 5.14 Proportion of water samples with detectable *Pseudomonas aeruginosa* (S = number of swimmers/100 m of beach)

The hypothetical curves in Figure 5.14, based on consultation with the Microbiology Component during the model-building workshops, are used to calculate the proportion of samples containing bacteria on the day of the month with maximum swimmers. To represent the increased growth of bacteria at higher temperatures, this calculated value is raised by five percent for each degree above 20°C. Referring to Figure 5.16, for example, with 60 swimmers per 100 m and no source of bacteria within 50 m, the predicted proportion of samples with detectable levels of *P. aeruginosa* is 0.3 or 30%. For a water temperature of 24°C the proportion would be raised by 20% (4 × 5%). Therefore, the proportion of positive samples would be 0.3 × 1.2 = 0.36.

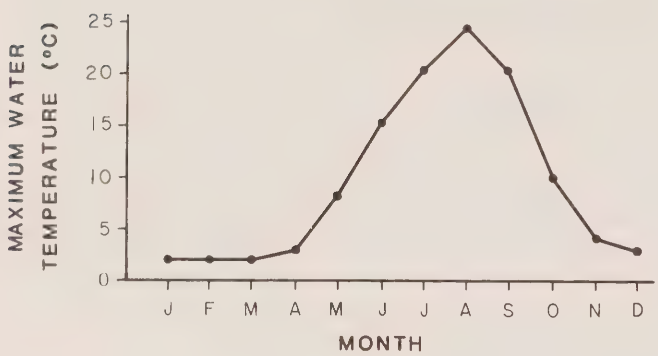


Figure 5.15 Maximum mean daily water temperatures at beach

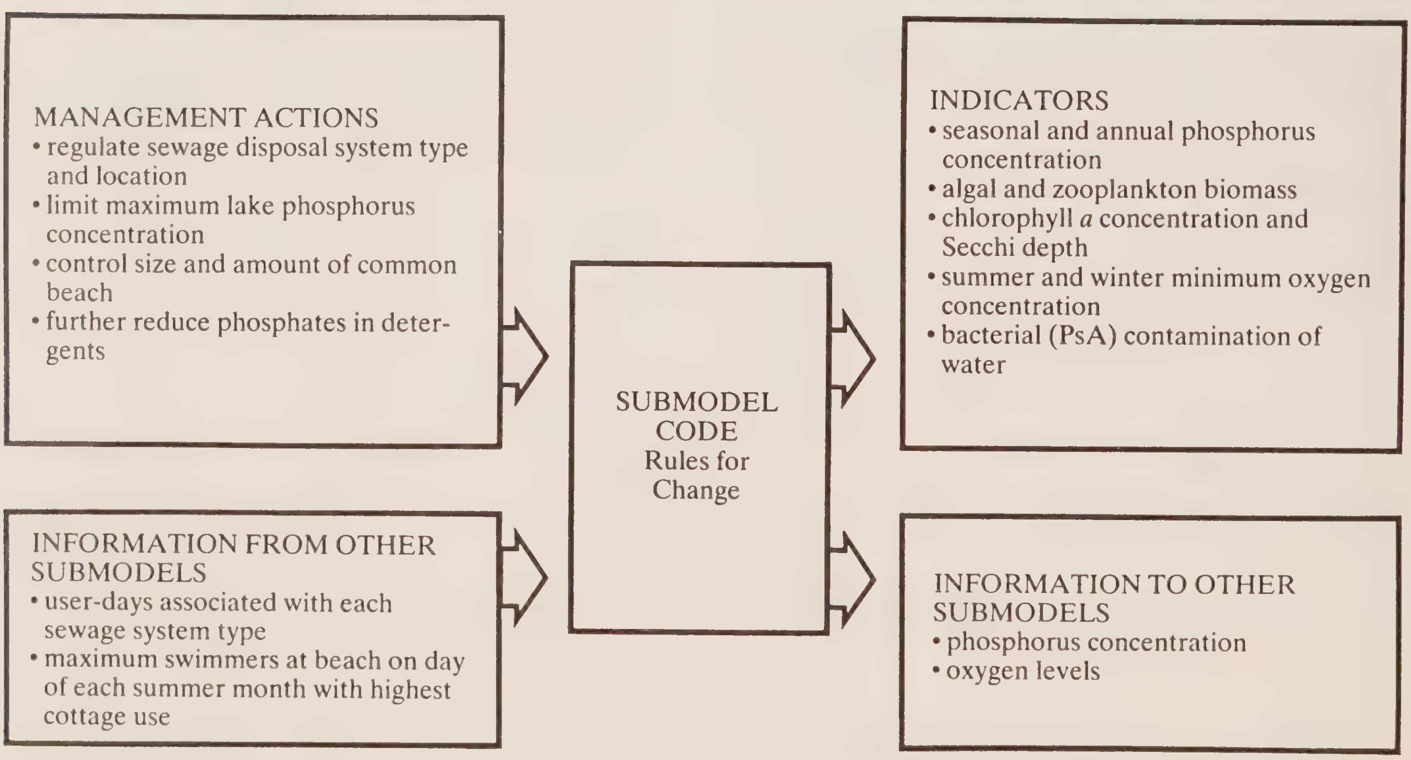
If the temperature drops below 10°C, the proportion of samples with bacteria is set to zero. Figure 5.15 is used to generate maximum mean daily water temperatures at the beach.

The prediction of the actual percentage of swimmers with ear infections was too uncertain to be included in the model. This uncertainty is primarily due to the difficulty of attributing recorded ear infections to bacteria in the lake where a swimmer is interviewed. Swimmers may have contracted the infection at other lakes or from a non-aquatic vector (Burger, pers. comm. 1981).

If a common beach is proposed, the user should specify the beach length which will then be associated with the development. If there is no public beach, an input of zero beach length initiates an algorithm to calculate frequency of occurrence of the bacteria along the shoreline, i.e. conditions likely to be encountered by cottagers swimming in the lake near their cottages (section 4.6).

5.14 SUMMARY

The Water Quality Submodel provides predictions of the impact of cottage development on the lake trophic status and microbiology. The submodel simulates the impact of existing and proposed cottages and can be altered in accordance with user-specified management actions. Information computed in the Land Use Submodel and relevant input data are used in the Water Quality Submodel to generate both the water quality indicators and specific information required by the Fisheries Submodel. The following input-output box diagram summarizes these submodel elements.



6. FISHERIES SUBMODEL

Fisheries Component scientists focussed their research on the relationship of angling activity to the depletion of the lake trout population in Study Area lakes (McCombie 1983). The researchers developed a set of linked equations which in total are able to produce predictions of:

a) the sport fish production capacity of a lake;
b) the amount of fishing effort by cottagers;
c) the amount of fishing effort by non-cottagers;
d) the seasonal (summer-winter) distribution of fishing effort;
e) the harvest resulting from the estimated total fishing effort; and
f) the relationship of the estimated catch to the amount that can be harvested indefinitely (surplus production) without adversely affecting the population.

The Fisheries Submodel developed for the OLCSM incorporates most of the Fisheries Component's research results. However, during the submodel development the boundaries were broadened to include the following:

a) a representative warm water species, the smallmouth bass;
b) the effect of water quality changes on fish production and survival; and
c) a linkage between the Land Use Component's prediction of cottage use and the calculated fishing effort.

The smallmouth bass was chosen because:

a) it does not compete for the same resource base as lake trout and should therefore be sensitive to different impacts of cottage development;
b) it is important to the recreational fisheries of Ontario Precambrian Shield lakes which include warm water lakes; and
c) data on smallmouth bass are available in the Fisheries Component data base and in the scientific literature.

A realistic measure of the cottaging impact is impossible without including the water quality effects on fish, e.g. reduction in hypolimnetic oxygen concentrations. For this reason, an oxygen linkage was created between the Water Quality and Fisheries submodels.

Finally, an additional equation for estimating fishing effort was included. This equation uses Land Use Component cottage-use data rather than the number of cottages. The advantage of this approach is that the fishing effort estimate reflects the lake-to-lake variations in cottage use resulting from differences in the type of cottage access. Presentation of both methods indicates the underlying factors involved and bares the strengths and weaknesses of each method.

6.1 PREDICTING FISH PRODUCTION

The core of the Fisheries Component study is the morphoedaphic index (MEI) developed by Ryder (1965). The index is calculated by dividing the total dissolved solids (TDS) measurement by the lake mean depth. It has been used as a rapid method of calculating potential fish yield for inland lakes in north temperate (Ryder 1982), subtropical and tropical (Schlesinger and Regier 1982) zones of the world. When Ryder regressed fish yield on the MEI the following predictive equation emerged:

$$MSY_c = 1.30 * MEI^{0.447} \qquad (6.1)$$

where MSY_c = maximum sustained yield of the lake's sport fish community ($kg \cdot ha^{-1} \cdot yr^{-1}$); and
MEI = morphoedaphic index (TDS/mean lake depth).

The assumption inherent in using the MEI is that the fish population is in equilibrium with both the biological environment and fishing pressure. These assumptions would be violated if fishing pressure were to drive fish populations to levels far below their equilibria or if, alternatively, acid rain were to raise total dissolved solids but directly deplete fish stocks. Therefore, in this submodel MEI is used as an indicator of potential production and linked to a more dynamic representation of fish population changes.

The basic formula used is the Schaefer equation (Schaefer 1957) for surplus fish production. The equation predicts the change in fish biomass (dB) with a change in time (dt):

$$\frac{dB}{dt} = RB\left(1 - \frac{B}{K}\right) - qEB \qquad (6.2)$$

where R = intrinsic growth rate of the population, also called natural growth rate (yr^{-1});
B = biomass of the population (kg wet weight $\cdot ha^{-1} \cdot yr^{-1}$);
K = carrying capacity of the environment for the species (kg wet weight $\cdot ha^{-1} \cdot yr^{-1}$);
q = catchability coefficient (Section 6.6); and
E = fishing effort (angler-hours $\cdot ha^{-1} \cdot yr^{-1}$).

This equation assumes that the growth rate of a fish population can be represented by a logistic curve (Figure 6.1). The curve shows three stages in the maturation of a population; the lower half of the curve rises rapidly, suggesting initial growth with little competition, followed by a dampening due to competition for food and space. Finally, a flat steady state is reached at which point growth and mortality are equal. Now biomass remains constant relative to the carrying capacity (K) of the lake.

In order to obtain the net biomass, the harvest term (q * E * B) is subtracted from the yield estimate (Equation 6.2). Harvest is estimated from the fishing effort and the proportion of the biomass caught in one hour of angling effort. The uncertainties associated with the determination of the latter (q) are discussed in Section 6.6. The effects of cottage development on lake trout and smallmouth bass are described as the changes in the values of R, K and E.

The Fisheries Submodel flow chart (Figure 6.2) indicates the calculations that are performed and their sequencing in the submodel. Provision has been made in the submodel for future inclusion of the linkage between changes in the littoral zone and smallmouth bass carrying capacity.

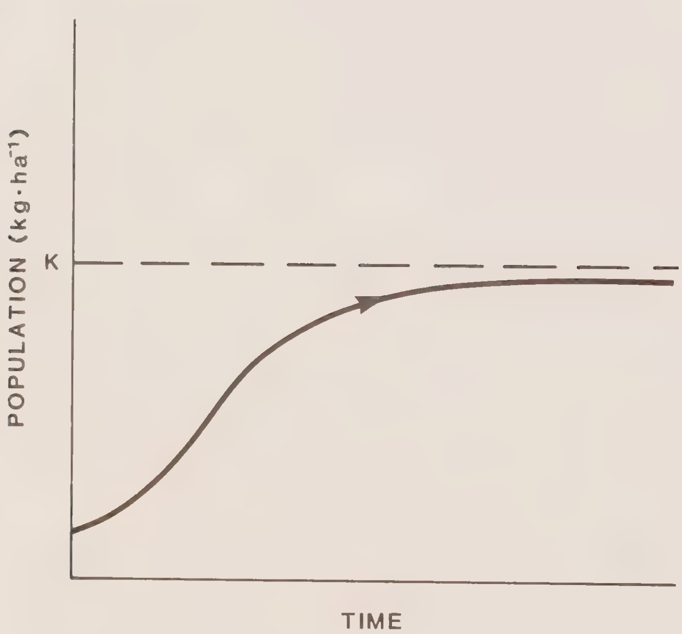


Figure 6.1 The logistic growth curve where K equals the population carrying capacity

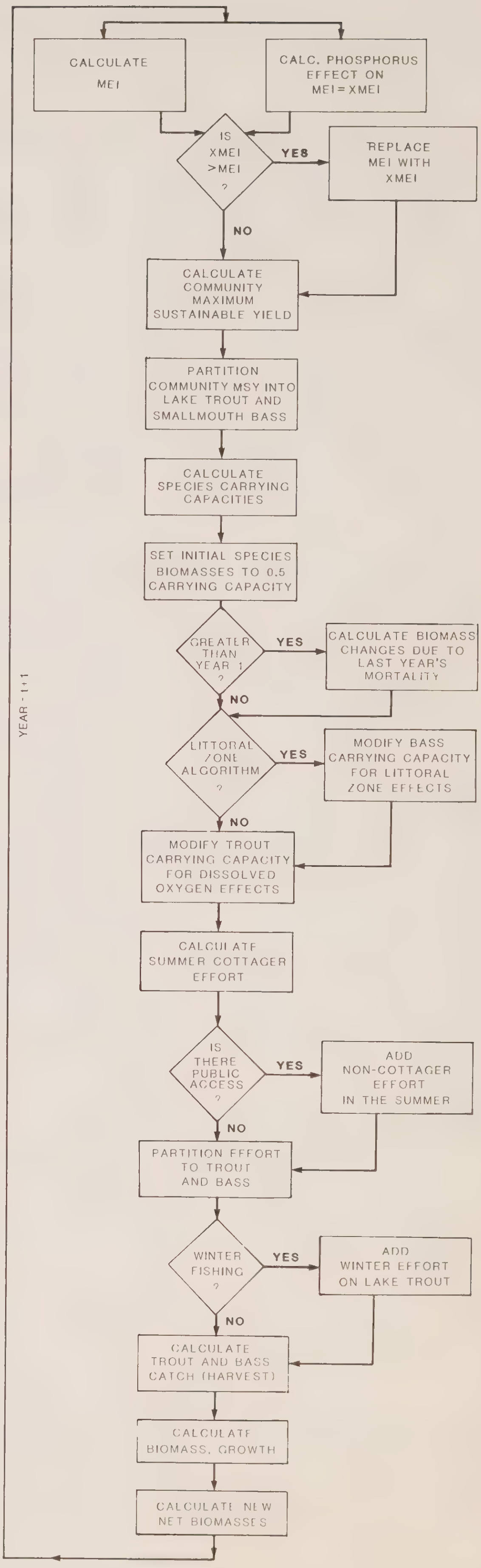


Figure 6.2 Fisheries Submodel flow chart

6.2 POPULATION GROWTH

It is assumed that the predicted yield from the MEI is equal to the maximum sustainable yield (MSY) (McCombie 1983). The biomass that yields MSY_c is assumed to be K/2 or one-half the environmental carrying capacity. If B is defined as the biomass at MSY_c:

$$\frac{K}{2} = B \text{ at } MSY_c \qquad (6.3)$$

and K/2 is substituted in Equation 6.2, then:

$$\frac{dB}{dt} = \frac{RK}{4} - qEB \qquad (6.4)$$

At equilibrium dB/dt = 0. Therefore:

$$\frac{RK}{4} = FB \text{ at } MSY_c \qquad (6.5)$$

where F = qE, the proportion of the fish biomass harvested, i.e. the instantaneous harvest rate.

At equilibrium, the total annual survival rate of a population can be estimated by (Ricker 1975):

$$e^{-Z} = \frac{MA}{1 + MA} \qquad (6.6)$$

where e^{-Z} = total annual survival rate;
Z = the instantaneous total mortality rate (natural plus harvest);
MA = mean age of the catch minus the youngest age at which fish are caught; and
e = 2.718, the base of the natural logarithm.

Given an estimate of the instantaneous natural mortality rate (M), the instantaneous harvest rate (fishing mortality) is:

$$F = Z - M. \qquad (6.7)$$

Combining B = K/2 and MSY_c = FB:

$$K = \frac{2 * MSY_c}{F} \qquad (6.8)$$

and because MSY_c = RK/4 (the growth element in Equation 6.4) the natural growth rate R becomes:

$$R = \frac{4 * MSY_c}{K} \qquad (6.9)$$

6.2.1 LAKE TROUT

The data in Martin and Fry (1973) were used to derive parameter values for lake trout population growth:

a) natural mortality (M) is 0.25 per year; and
b) the mean age of lake trout caught is 7 years and only fish 5 years and older are caught; therefore, MA = 7-5 = 2 years.

Therefore, from Equation 6.6, Z = 0.405 and from Equation 6.7, F = 0.16. The MSY calculated using MEI is for the entire sport fish community. Fisheries Component data (McCombie 1983) suggested that in absolute terms about 40% of this MSY would be in lake trout. Therefore, substituting 0.40 into Equation 6.9:

$$TR = \frac{4 * (0.4\ MSY_c)}{5 * MSY_c} = 0.32 \text{ year}^{-1} \qquad (6.10)$$

and by rearranging Equation 6.9:

$$TK = \frac{4 * (0.4\ MSY_c)}{0.32 \text{ year}^{-1}} = 5 * MSY_c \qquad (6.11)$$

where TK = trout carrying capacity; and
TR = natural growth rate of trout population.

As the MSY_c is for a fish community, a method was sought for more accurately apportioning a fraction of the MSY_c to lake trout. It was decided that the more than three decades of creel census observations on Lake Opeongo (Martin and Fry 1973) would be used as a guide for partitioning the MSY_c. For Lake Opeongo, the MEI and MSY_c are 1.57 $mg \cdot L^{-1} \cdot m^{-1}$ and 1.72 $kg \cdot ha^{-1} \cdot yr^{-1}$, respectively (MNR 1982b, McCombie 1983). As the yield has changed little in the past 15 years, one can assume that it approximates MSY_c. The lake trout yield from Lake Opeongo is 0.4 $kg \cdot ha^{-1} \cdot yr^{-1}$, therefore:

$$TMSYP = (0.4/1.72)MSY_c = 0.23MSY_c \qquad (6.12)$$

where TMSYP = the proportion of the MSY_c apportioned to lake trout.

This value ($0.23MSY_c$) has also been interpreted as the optimum sustained yield of lake trout for lakes in the Study Area.

6.2.2 SMALLMOUTH BASS

The data used to derive parameters of population growth for smallmouth bass came from various sources. The Ministry of Natural Resources (Shuter, pers. comm. 1981) provided most of the details of smallmouth bass biology needed to estimate BR and BK.

The fecundity of smallmouth bass is approximately nine times that of lake trout and smallmouth bass mature at a much younger age than lake trout (Scott and Crossman 1973). It was therefore hypothesized that the natural growth rate for smallmouth bass (BR) is higher than for lake trout. BR was set at 0.5 $year^{-1}$. Based on a Strategic Plan for Ontario Fisheries (SPOF) report (MNR 1982b), 50% of the MSY_c was assigned to smallmouth bass. Therefore, by rearranging Equation 6.9:

$$BK = \frac{4 * (0.5\ MSY_c)}{0.5 \text{ year}^{-1}} = 4 * MSY_c \qquad (6.13)$$

6.3 PHOSPHORUS ENRICHMENT AND THE MORPHOEDAPHIC INDEX

Phosphorus enrichment has been identified not only as a root cause of lake water quality and fish habitat degradation but, during the early stages of enrichment, as a contributor to increased fish production through an enhanced food resource for predatory fish (Oglesby 1977b, Hanson and Legget 1982). Although there are no data specific to lake trout or smallmouth bass, the assumption is that they, being predatory species, are affected by this stimulus.

The fish food is not phosphorus but the edible zooplankton which, in turn, depends on phytoplankton as a food base. Phytoplankton consist of the edible algae and the non-edible blue-green algae. The ratio of nitrogen to phosphorus determines the amount of blue-green algae produced (section 5.10). When the N:P atomic ratio is $<$ 14:1 and the base P concentration is $\geq 10\ \mu g \cdot L^{-1}$, less

phosphorus will be converted into usable fish food, thereby reducing the beneficial effect of enrichment. The nitrogen balance equation (Teleki and ESSA 1982) was included in the model to provide an estimate of the N:P atomic ratio. This permits the downward adjustment of the usable phosphorus value when the N:P ratio drops below 14:1 and the base P concentration rises above 10 $\mu g \cdot L^{-1}$. Although no lake in the Study Area has an N:P atomic ratio less than 14:1, it is important to account for this limnological relationship. In the present version of the model, the ratio is provided as an indicator and is not linked to the Fisheries Submodel because of the decision to calculate fish production from its relationship with lake nutrients (P and MEI) rather than from the linkage of P-plankton-fish production. Total nitrogen is given a mean default value of 664 $\mu g \cdot L^{-1}$.

At the same time that the "P-stimulated" total fish carrying capacity is rising, the habitat for selected sport fish, such as lake trout, is being stressed. For salmonids, in general, there is a fine line between the beneficial effect of enrichment and the negative effect of habitat degradation.

In the model, the positive effect of nutrient enrichment (phosphorus) is added to the existing MEI formulation in terms of a direct change in the MEI. The negative effect is applied by reducing the lake trout carrying capacity as a result of lowered hypolimnetic oxygen concentrations.

Data from 26 lakes (McCombie, pers. comm. 1982) were used to derive a relationship between phosphorus and the MEI.

$$XMEI = 178{,}595.59 * (YCPLK/1000)^{2.3551} \qquad (6.14)$$

where XMEI = the modified morphoedaphic index ($mg \cdot L^{-1} \cdot m^{-1}$);
YCPLK = the mean phosphorous concentration during the ice-free period ($\mu g \cdot L^{-1}$); and
$n = 26$
$p = <0.001$
$r^2 = 0.79$

As this regression equation proved the least accurate for small phosphorus values (actually depressing the modified MEI below the standard MEI), the submodel was redesigned so that XMEI would only be used if it was larger than the MEI. As the standard MEI is virtually a constant, the XMEI comes into use only when the phosphorus concentrations begin to rise.

6.4 DISSOLVED OXYGEN LEVELS AND LAKE TROUT

The growth and production of the lake biota is stimulated by nutrient enrichment of the lake, which is followed by increased oxygen consumption as the aquatic organisms die and decompose. Changes in dissolved oxygen (DO) levels can be linked to enrichment. During the submodel development, DO concentrations in lake hypolimnia were identified as being vital to the survival of lake trout, as they spend their entire lives in this region of the lake and must have well-oxygenated water. Smallmouth bass stay above the hypolimnion most of the time and are not affected until highly eutrophic conditions occur.

Generally speaking, lake trout are not affected by DO levels above 5 $mg \cdot L^{-1}$. At 2 $mg \cdot L^{-1}$ no lake trout can survive. The relationship over the range from 2 to 5 $mg \cdot L^{-1}$ is interpreted as a straight line (Figure 6.3).

Because no method is available for predicting hypolimnetic oxygen levels by depth stratum (hypolimnetic oxygen usually drops with depth), a conservative approach to oxygen forecasting was devised. The minimum oxygen concentration during the summer was used for all fish survival calculations. As changes in hypolimnetic DO are related directly to the trout carrying capacity (K), when DO drops below 5 $mg \cdot L^{-1}$ a progressively smaller portion of the initial carrying capacity (K) remains (Figure 6.3).

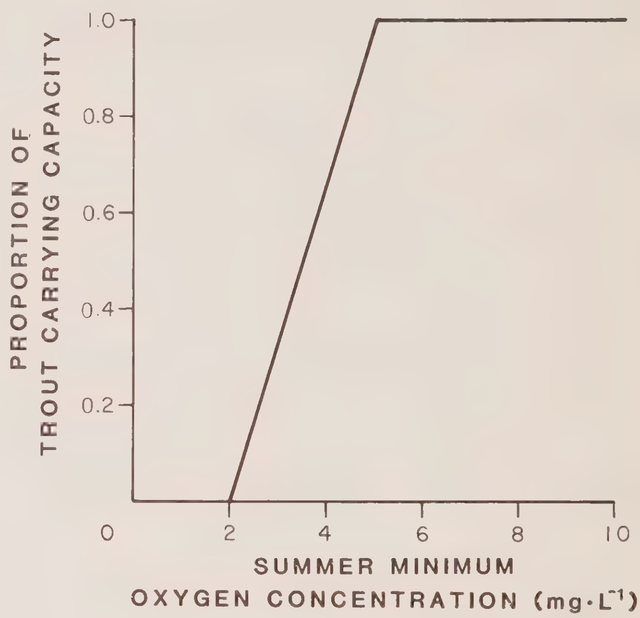


Figure 6.3 Effect of dissolved oxygen level in hypolimnion during the summer on lake trout carrying capacity

The mathematical calculation is simply:

TKO = TK * Proportion of initial TK remaining (from Figure 6.3)

where TK = the initial carrying capacity for lake trout derived from the morphoedaphic index; and
TKO = the potential lake trout K with the dissolved oxygen effect subtracted.

This is a powerful relationship since salmonid carrying capacity (K) and biomass are directly reduced once hypolimnetic oxygen drops below a mean of 5.0 $mg \cdot L^{-1}$.

The XMEI and oxygen effect have been programmed into the model so they always work together. If the nutrient enrichment causes depressed oxygen levels in the lake, then the lake trout carrying capacity is reduced proportionally at a rate shown in Figure 6.3. The model makes these adjustments during each simulation time-step.

As there are other stresses, particularly over-fishing, influencing lake trout production, being able to separate the enrichment effects (both positive and negative) from the fishing and other stresses, helps to identify the relative contribution of each stress to the changing fish production level.

6.5 FISHING EFFORT

A major effect of increased cottage development on fish populations is a rise in the amount of fishing effort. The Fisheries Component scientists addressed this problem by examining the following relationships:

a) cottager effort in summer as a function of the number of cottages and the lake surface area;
b) non-cottager effort in the summer as a function of summer cottager effort (provided that public access exists); and
c) total winter fishing effort as a function of total summer fishing effort.

6.5.1 COTTAGER FISHING EFFORT IN SUMMER

Summer fishing effort by cottagers was derived, initially, with an equation developed by McCombie (1983):

$$E_c = [(e^{1.576-0.000148 \cdot A}) * N] / A \qquad (6.15)$$

where E_c = cottager fishing effort in summer fishing season (angler-hrs · ha^{-1});
A = lake surface area (ha); and
N = number of cottages.

This equation is responsive to variations in fishing effort resulting from differences in lake surface area.

An alternate formulation was prepared using several Study Area and lake-specific constants in addition to a cottage use variable.

The advantage of the following equation is that it takes into account differences in cottage use from lake to lake, resulting from variations in the type of access, rather than assuming average cottage use on all lakes. Also, it allows the user to define the summer fishing season (which may vary locally) and to test management options by predicting angling effort under changed season lengths.

$$ECS = \frac{FUSES * PFISH * ANGLER * HOURS * PTIME}{AREAL} \qquad (6.16)$$

where ECS = cottager fishing effort in summer (angler-hours · ha^{-1});
FUSES = number of cottage-days during summer fishing period (May-September);
PFISH = proportion of cottages with occupants who fish in summer (= 0.5);
ANGLER = number of people per fishing trip (= 2);
HOURS = number of hours per fishing trip (= 2);
PTIME = proportion of total summer cottage-days spent fishing (= 0.33); and
AREAL = surface area of the lake (ha).

At present, both equations are programmed in the model but they operate independently. Further work is required to combine into one equation the strengths of the two methods of predicting summer cottager fishing effort.

6.5.2 NON-COTTAGER FISHING EFFORT IN SUMMER

McCombie (1983) regressed non-cottager effort in the summer on cottager effort in summer, as the creel census data indicated a relationship between the number of cottager and non-cottager anglers on the Fisheries Component study lakes. This regression equation is used in the Fisheries Submodel:

$$ENS = 0.518 * ACCESS * ECS^{1.2004} \qquad (6.17)$$

where ENS = non-cottager effort in the summer (angler-hours · ha^{-1}); and
ACCESS = a code to indicate whether there is a public access point on the lake. If ACCESS = 0, there is no public access. If ACCESS = 1, there is a maintained public access point and therefore non-cottager fishing.

6.5.3 APPORTIONING FISHING EFFORT

The proportion of fishing effort expended on the lake trout and smallmouth bass was initially divided equally between the two species. However, it remains a user-defined parameter. For example, field biologists may have first-hand knowledge of the effort allocation, in which case the new proportions would replace the old. Provision for a proportion of effort for other fish species has been made, as in some cases additional species make up a significant part of total fishing activity. The apportioning of effort is achieved with the following equations:

$$BEFF = 1.0 - TEFF - POEFF \qquad (6.18)$$

where BEFF = the proportion of the estimated fishing effort (angler-hrs · ha^{-1}) going to smallmouth bass;
TEFF = the proportion of the fishing effort going to lake trout, with data provided by the user; and
POEFF = the proportion of the total estimated fishing effort going to other species, with data provided by user.

6.5.4 WINTER FISHING EFFORT

McCombie (1983) hypothesized that, when selecting a winter fishing site, anglers relied on the fishing reputation of a lake. Waters with good summer lake trout fishing received higher winter fishing effort than lakes with a poor summer lake trout catch. McCombie developed the following regression equation for predicting winter lake trout fishing effort:

$$EFFW = 0.811 * WINTER\ (EFFS)^{0.962} \qquad (6.19)$$

where EFFW = winter fishing effort for lake trout (angler-hours · ha^{-1});
EFFS = summer fishing effort by cottagers and non-cottagers (angler-hours · ha^{-1}); and

WINTER = a switch to mimic presence or absence of winter fishing (not part of McCombie's equation). If WINTER = 0, there is no winter fishing. If WINTER = 1, there is winter fishing.

Data collected by both the Fisheries and Land Use Components indicate that winter anglers are largely non-cottagers. The Land Use Component Survey of Lakeshore Residents revealed that although 49% of the cottages have occupants who fish in summer, only 5% of the cottages have occupants who fish in winter (MOH 1979). This is important when evaluating management options, as reducing the number of cottages will have only a slight effect on winter fishing effort.

6.5.5 FISHING EFFORT CONTROLS: SHORTENING SEASONS

When faced with dwindling fish stocks and a growing angler population, managers sometimes shorten the fishing season. This has the effect of reducing the total harvest, thus allowing more fish to mature, reproduce and replenish the population. In the model, the user can 'set' the fishing season by giving new numerical values to the following variables:

- ISS — first month of summer fishing season;
- ISF — last month of summer fishing season;
- IWS — first month of winter fishing season;
- IWF — last month of winter fishing season; and
- ISEAS — a switch which turns the entire summer fishing season on (ISEAS = 1) or off (ISEAS = 0).

For example, ISS = 7 and ISF = 8, sets the summer fishing season to two months, July and August; or IWS = 1 and IWF = 3 sets the winter season to January through March.

6.6 CATCHABILITY COEFFICIENT

The catchability coefficient "q" (Equation 6.20) is the proportion of the fish stock caught per unit of fishing effort, e.g. boat-day, rod-hour or angler-hour. The literature is replete with analyses (Peterman and Steer 1981, Paloheimo 1980, Holling 1959) suggesting how the q is related to fish abundance, how it should be derived and ultimately how elusive a parameter (or variable) it can be. Using Holling's work (1959), Peterman and Steer (1981) described the relationship of q to fish abundance and suggested that an exponential decay curve (Figure 6.4, A) was the most appropriate for fish harvest situations — as the fish population grows, q drops.

What clouds this relationship is the influence of angler skill and gear efficiency on q. Unfortunately, there are no functional relationships that can be applied to correct for this factor, so the best available solution is to make q as lake-specific as possible.

6.6.1 LAKE TROUT q

The steps leading to a lake-specific lake trout q began with a conversion of Paloheimo's (1980) q for Lake Opeongo lake trout from 0.0262 per hundred boat-hours to 0.000131 per angler-hour. In applying this q, few if any trout populations were ever stressed, even with unreasonably high levels of fishing effort.

Since q is influenced by fish abundance which, in turn, is a function of the trophic status of the lake, and since abundance is also related to fishing pressure, a forward stepwise regression analysis was performed. This was done to determine the combination of fishing effort, MEI and/or biomass which was the most useful in predicting q.

The q values for the regression were derived from a reformulation of the harvest (Y) part of the Schaefer equation (Equation 6.4). If $Y = qEB$, then $q = Y/(E \times B)$. The lake trout summer effort and harvest data were obtained from the Fisheries Component Creel Census. The predicted lake trout biomass was used because there were no measured values available. The biomass prediction represents an estimate of biomass at the maximum sustained yield.

The analysis, using Fisheries Component lakes, revealed that, for lake trout, fishing effort and MEI were the best predictors of q and that these two variables did not exhibit multicollinearity. The intention was that this formulation be used in the model and the method of estimating q re-evaluated following the collection of new q related fisheries data. The resulting equation, using the SAS (1979) STEPWISE procedure was:

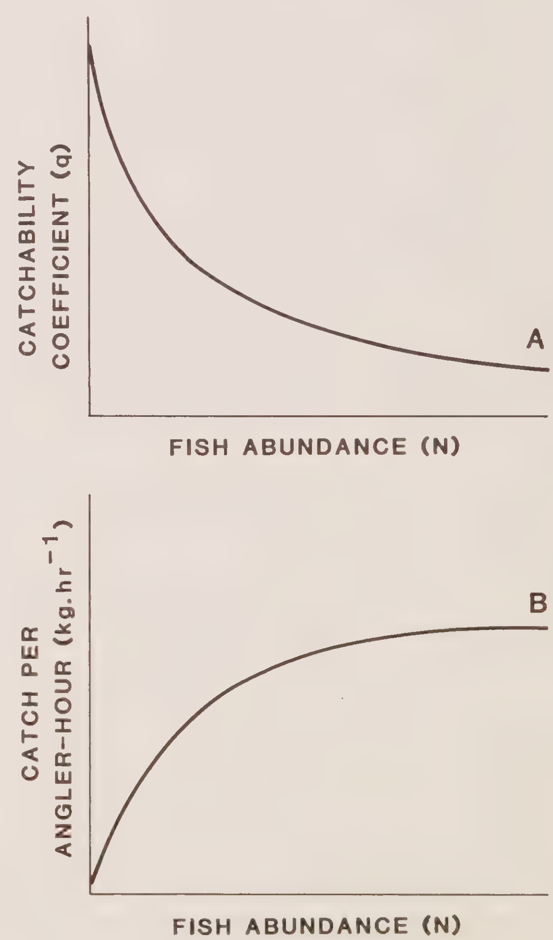


Figure 6.4 Most likely response of the catchability coefficient (A) and catch (B) to changes in fish abundance, e.g. biomass (from Peterman and Steer 1981 after Holling 1959)

$$q_{LT} = 0.642 - 0.0365\,(\ln MEI) + .0011\ \text{Effort}_{LT} \qquad (6.20)$$
$$n = 12$$
$$F_{0.023} = 6.32;\ 2, 8\ \text{d.f.}^{1}$$
$$r^2 = 0.61$$

where

q_{LT} = catchability coefficient for lake trout;
Effort_{LT} = total summer effort for lake trout; and
ln = natural logarithm

The fact that effort is implicit in q strengthens the argument that it be applied to calculate lake-specific q values. Both variables reflect the response curve established by Peterman and Steer (1981) in that, as effort rises, catch increases. The resulting reduction in fish abundance causes q to rise. Similarly, as MEI rises fish abundance tends to rise, i.e. MEI is indirectly related to q (correlation coefficient = – 0.64).

[1] "F" statistic is the ratio of explained variance to unexplained variance. The larger the "F" statistic, the smaller is the unexplained variance portion of the statistic in relation to the explained portion.

The q values for summer and winter fishing are assumed to be equal as no data are available substantiating a consistent difference in the q between fishing seasons (McCombie 1983).

6.6.2 SMALLMOUTH BASS q

Since Lake Opeongo data did not include the q for smallmouth bass, a different approach was taken. The basic assumptions used in calculating smallmouth bass q were:

a) The natural mortality rate for smallmouth bass is 0.25 per year, minimum estimate (Shuter, pers. comm. 1981)
b) Approximately 50% of the MSY_c estimated harvest is smallmouth bass (MNR 1982)
c) The fecundity of smallmouth bass is about nine times that for lake trout. Also, the age at first maturity for smallmouth bass is less than that for lake trout (Scott and Crossman 1973). Therefore, it was hypothesized that BR was about 70% higher than TR. As a trial value, BR was set at 0.54 yr^{-1} ($1.7 * TR = 0.54\ yr^{-1}$)
d) Using equation 6.4 and substituting for BR: $BK = 3.7\ MSY_c$
e) Catch per unit of effort (CPE) for lake trout is 3.1 times that for smallmouth bass. From the Fisheries Component study data (McCombie, pers. comm. 1981) on 12 lakes, $CPE_{SMB} = 12.5$ $gm \cdot angler\text{-}hour^{-1}$ and $CPE_{LT} = 38.8$ $gm \cdot angler\text{-}hour^{-1}$.

Since: $CPE = qB$

and the average biomass of lake trout is approximately six times the average biomass of smallmouth bass, then:

$$\frac{q_{LT}}{q_{SMB}} = \frac{38.8}{12.5} * \frac{B_{SMB}}{6 * B_{SMB}}$$

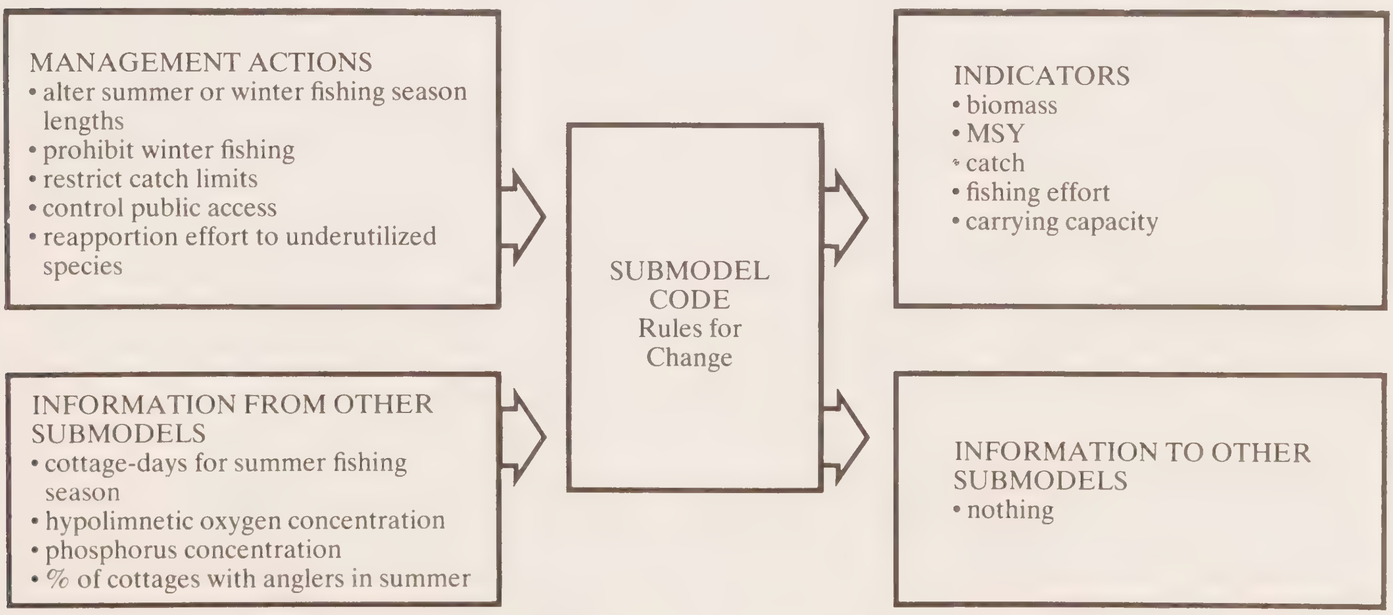
where q_{LT} = as above.

Solving for q_{SMB} produces the following equation:

$$q_{SMB} = \frac{q_{LT} * 6B_{SMB}}{3.1\ B_{SMB}} \qquad (6.21)$$

6.7 SUMMARY

The Fisheries Submodel provides predictions of the impact of cottage development on the sport fish population of a lake. The submodel simulates the impact of existing and proposed cottages and can be altered in accordance with user-specified management actions. Information computed in the Land Use and Water Quality submodels and relevant input data are used in the Fisheries Submodel to generate fisheries indicators. The following input-output box diagram summarizes these submodel elements.



7. WILDLIFE-HABITAT SUBMODEL

The basic underlying assumption of the Wildlife Component research is that lakeshore cottage development alters the structure and species composition of the shoreland vegetation, i.e. the wildlife habitat. The alteration of wildlife habitat, in turn, affects wildlife populations (Figure 7.1).

The Wildlife-Habitat Submodel first estimates an index of disturbance, based on the density of cottage development. This index is the basis for many of the other calculations in the submodel. A flow chart of the submodel is shown in Figure 7.2.

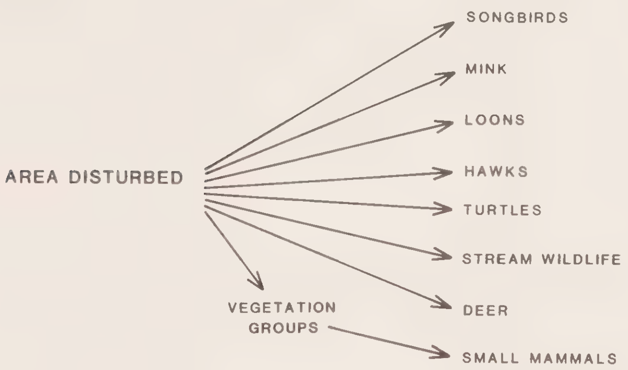


Figure 7.1 Impact of cottages on wildlife habitat for selected species, using area disturbed by development as the basis for evaluation

7.1 SPATIAL REPRESENTATION OF THE LAKESHORE AND LITTORAL ZONE

The area considered to be important to wildlife is a 50-m band of shoreland surrounding the lake. In order to relate the location of cottages to types of wildlife habitat, this strip of land is divided into segments of equal size. The user chooses a uniform segment width within the range of 50 to 200 m. The smaller segments are preferable as they best describe the amount of transition zone between developed and undeveloped sites. Each wildlife segment is classified into one of three forest cover types: coniferous, deciduous, or mixed (coniferous plus deciduous).

The lot size is the area of the lot within the 50-m strip of shoreland, i.e. the width of the registered lot multiplied by 50 m, rather than the area of the registered lot.

The littoral zone is divided into a linear grid of segments of the same width as the segments on the adjacent lakeshore. The distance they extend into the lake is not explicitly defined. The littoral zone segments are classified by type: weedy (includes litter) or rocky (includes sand and gravel).

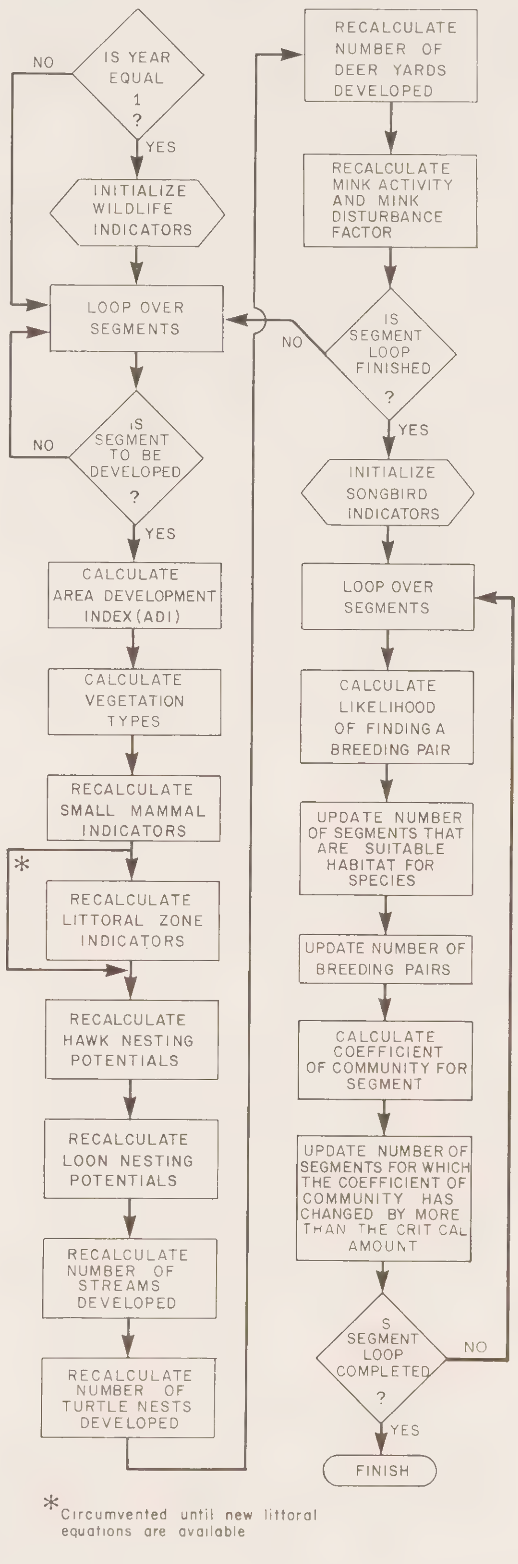


Figure 7.2 Wildlife Submodel flow chart

7.2 HABITAT DISTURBANCE

For many wildlife species, any degree of cottage development on a lakeshore segment makes it unsuitable as wildlife habitat. This is the case for deer, turtles and animals that rely on stream habitats. For others, such as small mammals, mink, songbirds, loons and hawks, it is the degree of disturbance that is important.

The degree of disturbance was calculated directly from measurements of the proportion of the area of the Wildlife Component study plots disturbed in each of three shoreland vegetation layers (ground, shrub, and tree). These three proportions were summed to obtain an index of disturbance which has a range in value from 0.0 to 3.0, with 0.0 representing minimum disturbance and 3.0 representing maximum disturbance. When applied to a cottage lot, it is termed the Area Development Index (ADI).

$$ADI = (Ag + As + At) / AREA \qquad (7.1)$$

where Ag = area disturbed (ha) in the ground layer due to cottage development;
As = area disturbed (ha) in the shrub layer due to cottage development;
At = area disturbed (ha) in the tree layer due to cottage development; and
AREA = area (ha) of the cottage lot (within the 50-m band).

The Wildlife Component team found that the actual area disturbed per lot was relatively uniform and independent of the size of the lot. As the lot size increased, the proportion of the lot disturbed decreased, leading to a reduction in the ADI (Figure 7.3).

In the Wildlife-Habitat Submodel, the ADI is calculated from the mean lot size per segment and is used to predict habitat disturbance for small mammals, songbirds and mink.

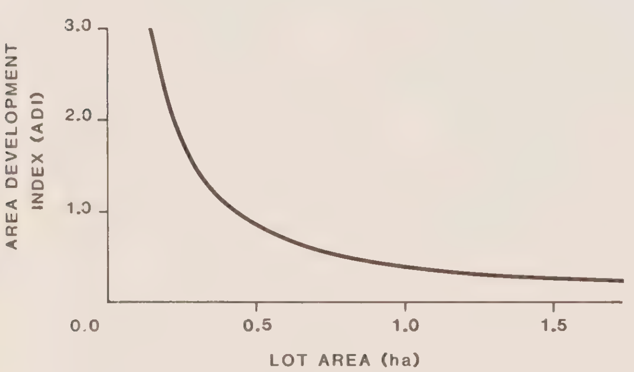


Figure 7.3 Area Development Index as a function of lot size

7.3 CAPTURE INDICES FOR SMALL MAMMALS

The Wildlife Component scientists (Racey and Euler 1982) were able to show that one could predict the change in small mammal numbers by examining changes in the amount of vegetation (single species or an assemblage of species) within each lakeshore segment.

Best fit regression equations were developed using several multivariate statistical procedures (Racey and Euler 1982), resulting in an equation which, for each major forest cover type, could best predict the relative small mammal populations. The Capture Index is an index of population size rather than absolute numbers for each species.

A statistical technique was used to select those vegetative groups which were most useful in the prediction. For example, Figure 7.4 illustrates the relationship used for estimating the density of the ground species *Clintoria borealis* (CLIN) in coniferous habitat at different levels of disturbance. Similar relationships were developed for 108 different vegetation groups occurring in the three major forest cover types (coniferous, mixed, deciduous). These relationships are internal to the submodel program.

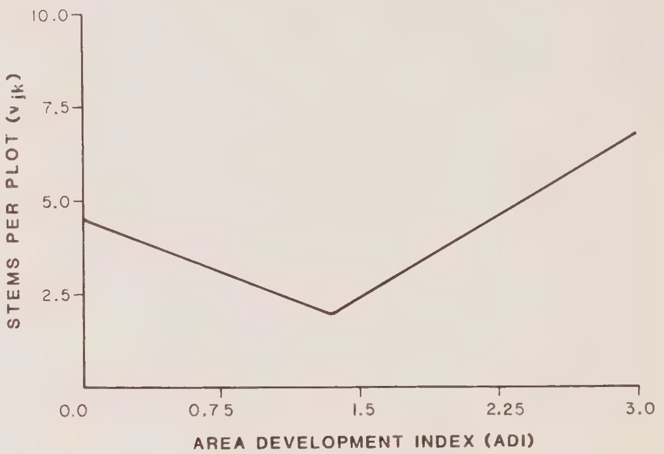


Figure 7.4 Relationship between density (stems/plot) of *Clintonia borealis* in coniferous habitat and the Area Development Index used in the submodel

The small mammal group in the Wildlife Submodel includes shrews (2 species), mice (2 species), chipmunks, squirrels and voles. The general form of the equation used in the submodel to calculate the Capture Index for each of the seven small mammal species is:

$$CI_{ij} = a_j + \sum_{k=1}^{n} b_{jk} * v_{jk} \qquad (7.2)$$

where CI_{ij} = Capture Index for small mammal species i in major forest cover j;
a_j = regression equation intercept for forest cover type j (deciduous, mixed or coniferous);
b_{jk} = regression coefficient associated with the major forest cover j and the species group k;
v_{jk} = estimate of amount of vegetation (stems/plot) for the specific species group k within the major forest cover j; and
n = maximum number of regression coefficients (i.e. b_{jk}) as determined by the regression analyses.

Several b and v values make up each regression equation. The predictive equation for the Capture Index of the masked shrew, *Sorex cinereus* in coniferous habitat is:

$$CI = -0.188 + 0.054(BETP) + 0.015(CLIN) \qquad (7.3)$$

where BETP and CLIN are the v_{jk} vegetation values

The capture indices for small mammals on individual shoreland segments are summed to obtain a lakewide index for measuring changes in small mammal habitat.

7.4 MINK

The submodel calculates two measures of the change in mink habitat: the lakewide mink activity index and a mink disturbance index.

7.4.1 MINK ACTIVITY INDEX

The Wildlife Component scientists developed an equation that gives a numerical value to the level of mink activity along the lakeshore, based on signs of mink activity observed in the field. The mink activity index (M) was calculated for each study plot and 50 m on either side of it in the following way:

$$M = S + P + D + T \qquad (7.4)$$

where S = log (no. of scats);
P = scent posts: either 1 for a minor scent post or 2 for a major scent post;
D = dens: either 2 for a possible den[1] or 5 for a confirmed den; and
T = tracks: 1 if the only sign of mink activity is a track.

In the Wildlife-Habitat Submodel, the mink activity index is calculated from its relationship with the forest cover type and ADI in each lakeshore segment (Figure 7.5). As cottage development increases, the submodel computes the change in the mink activity index from the index prior to the existing development (predevelopment).

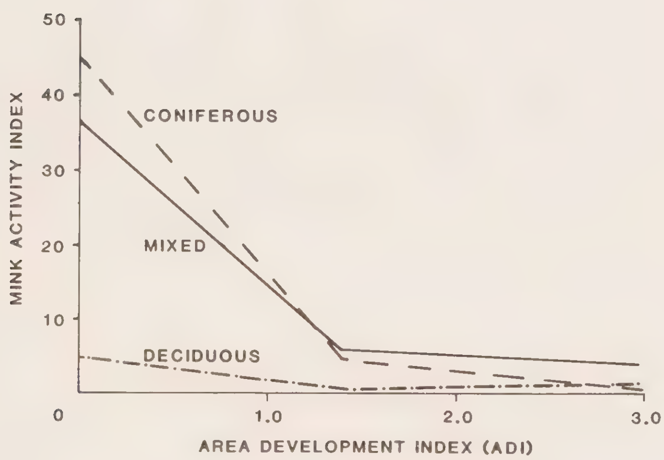


Figure 7.5 Mink activity index as a function of the Area Development Index, for three forest cover types

[1] A possible den was defined as lacking fresh scats on at least one of the shorewalking rounds while a confirmed den had fresh signs of mink activity on each round.

7.4.2 MINK DISTURBANCE INDEX

The mink disturbance index is a theoretical index, based on the proportion of the lakeshore that is developed and the level of human activity as measured by the amount of cottage use in July, the peak month for cottagers. It is assumed that as the amount of developed shoreland increases, as indicated by the number of developed segments, the degree of mink disturbance also increases. It is also assumed that the degree of disturbance is intensified by greater human activity which is associated with the higher number of user-days in July (Figure 7.6).

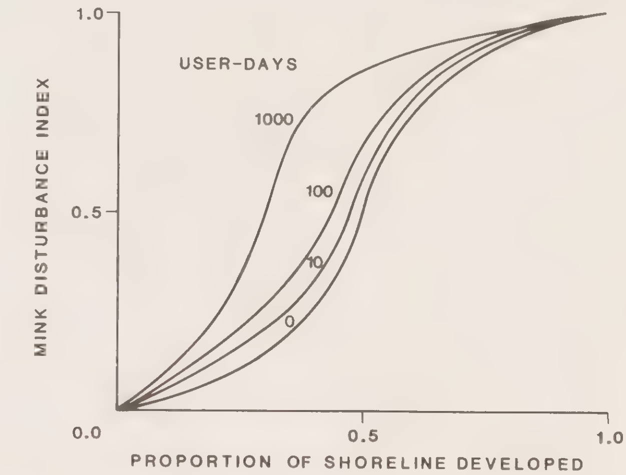


Figure 7.6 Mink disturbance index as a function of lakeshore cottage development and number of user-days during critical month (0 = shoreline developed but no occupants during the month)

7.5 SONGBIRDS

Songbird habitat is dependent upon forest cover type, which is classified as deciduous, mixed or coniferous. New forest edge, created by clearing vegetation for cottage development, changes the bird community structure and species composition. Whereas the robin can utilize a wide range of conditions from lawns to less disturbed forest habitat, other species, such as the red-eyed vireo, require mature trees.

The songbird portion of the model is based on the assumption that the probability of finding a breeding pair of birds on a given segment is related to the degree of disturbance on the segment and the degree of disturbance on adjacent segments. Shoreland segments for the songbird portion of the submodel extend inland 100 m, instead of 50 m, from the shoreline.

7.5.1 DETERMINING THE PROBABILITY OF FINDING A BREEDING PAIR OF BIRDS

For each species, the probability of finding a breeding pair of birds on a given segment depends on four factors: 1) the forest type, 2) the size of the segment, 3) the degree of development on that segment, and 4) the number of adjacent segments that are developed. Table 7.1 shows the likelihood of finding ovenbirds on a 100 ×

100 m segment. For example, on an undeveloped segment with deciduous forest and no adjacent segments developed, the likelihood of finding a pair of ovenbirds would be 0.6 or 60%. On a heavily developed segment with the same type of forest, the value is reduced to 0.2 or 20%.

Table 7.1 Likelihood of finding a breeding pair of ovenbirds on a 100 × 100 m segment, for different forest types and levels of disturbance

Area Development Index/6.0	DECIDUOUS Number of Adjacent Segments Disturbed			MIXED OR CONIFEROUS Number of Adjacent Segments Disturbed		
	0	1	2	0	1	2
0.0	0.6	0.5	0.3	0.9	0.7	0.5
0.100 – 0.349	0.3	0.3	0.3	0.5	0.5	0.5
0.350 – 0.500	0.2	0.2	0.2	0.4	0.4	0.4

The probabilities were determined by the research team and have been programmed into the submodel code for 19 songbird species. Included among the species are representatives of the flycatchers, thrushes, warblers and finches.

After the probabilities are calculated, the submodel compares them to a critical value for each species. If the likelihood is greater than the critical value, the segment is considered suitable and at least one breeding pair is added to the total expected number of breeding pairs for that species. If the likelihood is less than the critical value, the segment is assumed to be unsuitable. In this way, the number of suitable segments for each songbird species can be calculated. The lakewide number of breeding pairs of each species is the sum of the probabilities for a given species on all segments around the lake.

7.5.2 COEFFICIENT OF COMMUNITY

The Coefficient of Community (CC) or the Jaccard index (Jaccard 1932) measures the degree to which the species composition on a study plot resembles that of the most highly disturbed plot.

For each study plot a CC was calculated using the following equation:

$$CC = [c/(a + b - c)] * 100 \tag{7.5}$$

where CC = Coefficient of Community;
c = number of bird species common to both plots;
a = number of bird species in most highly developed plot; and
b = number of bird species on the sample plot.

The CC was broken down by the dominant forest cover types on the study plots, i.e. coniferous (including mixed plus coniferous) or deciduous and then related to the ADI/6 (Figure 7.7).

The resulting regression equations are:

$$CC_{conif.} = 65(ADI/6) + 14.9 \tag{7.6}$$

and

$$CC_{decid.} = 50(ADI/6) + 34.4 \tag{7.7}$$

The Coefficient of Community is estimated for each segment, using the regression relationships shown in Figure 7.7. The submodel calculates the proportion of change in the Coefficient of Community from its initial state, i.e. the native songbird community. The number of segments in which the change is greater than a critical value are summed. The total number of segments where the Coefficient of Community has changed by more than the critical value is a measure of how the songbird community around the lake has been altered.

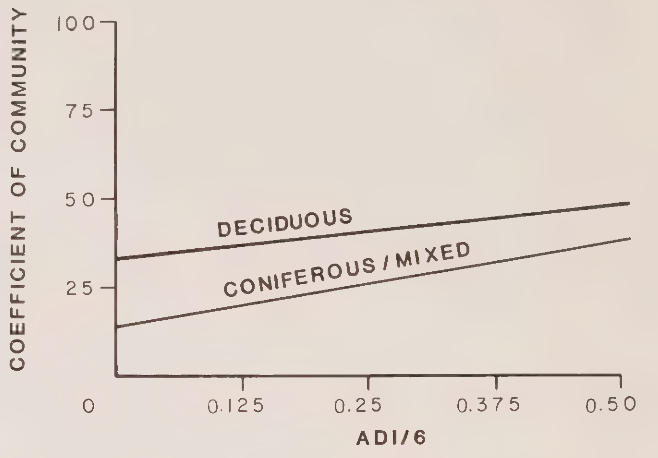


Figure 7.7 Regression lines for estimating the Coefficient of Community

7.6 LOONS

The location of each potential loon nesting site is stored in the model. It is identified by the lakeshore segment number in which the nesting site is located. In addition, the number of loon nesting islands are identified. Each of the shoreline loon nesting sites and loon nesting islands is assigned a loon nesting potential (a value between 0 and 1). Initially, each is given a value of one, the highest potential. When a segment is developed, the loon nesting potential for each shoreline nesting site is recalculated, based on the distance between the closest boundaries of the segment with the loon nest and the segment being developed (Figure 7.8). The distance is set to zero when the development and the loon nest occur on the same segment.

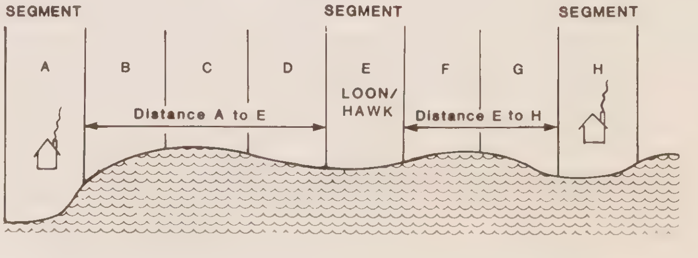


Figure 7.8 Distance measurements used in calculating loon or hawk nest disturbance from development on segments A and H

Once the distance to the nearest developed segment is determined, each potential nesting site is assigned a value. For distances equal to or greater than the safe distance, the loon nesting potential is not affected; for distances less than the minimum distance, the loon nesting potential is zero (Figure 7.9). When the nest is located between the minimum and safe distance from a developed segment, the nesting potential is set at a value proportional to its location between the designated minimum and safe distances.

When the minimum and safe distances are set at the same value (e.g. 250 m), the loon nesting potential will be zero if the distance is less than the specified value and one if the distance is greater than or equal to the specified value. On the recommendation of the Wildlife Component, both the minimum and safe distances are set at 250 m in the submodel.

Once the loon nesting potentials for all shoreline nesting sites are determined, they are added to the number of loon nesting islands to yield a loon nesting potential for the entire lake.

Figure 7.9 Relationship of distance (m) from the nesting segment to the nearest developed segment and the probability of successful loon and hawk nesting

7.7 HAWKS

The location of broad-winged hawk nesting valleys is also recorded in the model, by the segment numbers in which they are found. Each of the hawk nesting valleys is initially given a hawk nesting potential of one. This value is then recalculated based on distance, in a manner analagous to loon nesting sites. The distance is determined as in Figure 7.8. For distances greater than or equal to the safe distance, the hawk nesting potential is not affected; for distances less than the minimum distance, the hawk nesting potential is zero. Nests located between these two points are assigned a proportional value. On the recommendation of the Wildlife Component, both the minimum and safe distances for the broad-winged hawks are set at 150 m.

7.8 CRITICAL WILDLIFE HABITAT

The user supplies the submodel with the location of deer-yards, turtle nesting areas and streams, by segment number. The submodel records and supplies, as output, the number of designated segments with critical habitat that have been developed and when, during the simulation, this development took place.

The Wildlife Component research indicated that cottage development in winter deer concentration areas disrupted this habitat and reduced its suitability for deer. Natural turtle nesting sites should be preserved. Streams were also considered critical habitat, as they provide valuable nesting, feeding and shelter areas as well as travel lanes for many wildlife species.

7.9 SUMMARY

The Wildlife-Habitat Submodel provides predictions of the impact of cottage development on the lakeshore wildlife and wildlife habitat. The submodel simulates the impact of existing and proposed cottages and can be altered in accordance with a range of user-specified management actions. Information generated in the Land Use Submodel and relevant input data are used in the Wildlife-Habitat Submodel to produce indicators of change in the lakeshore wildlife community. The following input-output box diagram summarizes these submodel elements.

MANAGEMENT ACTIONS

- relocate proposed lots relative to nesting sites
- reserve lakeshore areas for animal habitat
- control minimum lot width
- vary acceptable levels of habitat change
- reduce clearing of vegetation on lots

INFORMATION FROM OTHER SUBMODELS

- user-days in July

SUBMODEL CODE
Rules for Change

INDICATORS

- Area Development Index
- disturbance indices for small mammals and songbirds
- mink activity and disturbance indices
- indicators of habitat loss
- development in critical areas

INFORMATION TO OTHER SUBMODELS

- number of cottages per subdivision in each year

8. FUTURE MODEL REFINEMENTS

The Ontario Lakeshore Capacity Simulation Model (OLCSM) is a simplified representation of a dynamic ecosystem. Only those elements that are necessary to simulate lake and lakeshore conditions are included. While the model does not provide answers to all the questions considered by planners in reviewing a subdivision proposal, it does substantially reduce the number of unknowns with respect to future environmental impacts.

8.1 KEY FEATURES OF THE PRESENT VERSION OF THE OLCSM

The OLCSM unites the various methods of prediction developed by the Lakeshore Capacity Study research teams, to provide a single practical tool for estimating the environmental change associated with cottage development on the lakeshore.

One of the advantages of linking these separate submodels into one integrated model is that it precludes using inconsistent data, which can occur if each submodel is operated independently. In the simulation model, all submodels draw data from the same input form, thus ensuring a consistent data base.

The environmental impact of lakeshore cottage development is measured in terms of a selected group of indicators, which are the most significant variables in the land-lake system for evaluating responses to development (Table 8.1). The values of these indicators describe the extent of the impact.

The model provides an objective estimate of environmental impact. It is not designed to make judgments about the numerical limits of the indicators. These are determined by the user, in the context of policy and technical guidance governing the particular situation.

A major feature of the model is its ability to respond quickly. This is possible because availability of data was one of the criteria applied in selecting the input requirements for each submodel.

Another practical advantage of the simulation model is its ability to test the effects of various management strategies. The user can introduce or terminate a particular strategy at any time during the simulation period, in order to determine whether a potential environmental problem can be solved by mitigative measures.

The OLCSM is designed to be flexible and easily modified. Experimental changes in the value of any variable are possible at any time during the simulation, as are modifications and additions to the model code. These various features create a powerful environmental planning tool.

Table 8.1 OLCSM indicators most sensitive to changes in the values of key variables and parameters

Submodel	Key Variables and Parameters	Most Sensitive Indicators*
Land Use	Number of cottages	Cottage-days
	Travel mode	User-days
	Cottage typology	
Water Quality	User-days	Mean ice-free period phosphorus concentration
	Lake surface area	Mean summer chlorophyll *a* concentration
	Phosphorus sedimentation rate	Hypolimnetic oxygen concentration
	Extent of spring and fall turnover	
Fisheries	Lake surface area	
	Mean ice-free period phosphorus concentration	Fishing effort (by species)
	Fish population growth rate	Fish harvest (by species)
	Catchability coefficient	Fish biomass (by species)
	Summer fishing-days	
	Fishing effort allocation by species	
Wildlife-Habitat	Lot area	Area Development Index
	Coniferous tree cover	Small Mammal Index (by species)
	Minimum and safe distance between potential nesting site and development	Mink Index
		Songbird Index (by species)
		Loon nesting potential
		Hawk nesting potential

* Based on Sensitivity Analysis (Appendix B)

8.2 AN EXAMPLE OF MODEL SIMULATION

The method of predicting, interpreting and mitigating the impact of cottage development on a hypothetical lake in the Muskoka-Haliburton area is illustrated in the example in Appendix C. The model was run for six different scenarios: first, with existing development; second, with proposed development added; and thereafter with selected mitigative measures. The predictions for selected indicators, shown on line graphs, are interpreted in the text.

Other examples could be used to demonstrate that the same number of cottages might have quite a different impact on another lake with different lake and watershed characteristics. In addition, if cottages are built on an upstream lake, they may increase the phosphorus concentration in the lake and in the outflow streams, thus possibly affecting the cottage capacity of the downstream lake.

The effects of variations in such factors as number and location of cottages, type of cottage access, timing of construction, lake and watershed characteristics, type of wildlife habitat and selection of management strategies can all be simulated by applying the OLCSM. The resulting predictions of probable impact can assist planners in guiding development in ways that will avoid damage to the inland lake environment.

8.3 RECOMMENDED FUTURE REFINEMENTS

The refinements specifically identified in the descriptions of the submodels (Chapters 4-7) should improve the model's predictive capability. Therefore, the next version of the OLCSM should include as many of these refinements as possible.

With respect to water quality, the new equations developed by the Trophic Status Component group, for gross phosphorus export and lake phosphorus retention, should be incorporated into the simulation model. Further, the prediction of depth-related minimum oxygen values for the hypolimnion would strengthen the linkage between the Water Quality and Fisheries submodels, because of its importance in defining fish habitat.

In the Fisheries Submodel, the method of estimating the catchability coefficient should be improved or replaced, as the fish harvest prediction depends heavily on this parameter. In addition, the choice between the two methods of predicting fishing effort needs to be resolved.

The quantitative linkage between cottage development and alterations in the littoral zone, coded in the Wildlife-Habitat Submodel, requires further work by the Ministry of Natural Resources. As the framework for including these relationships in the OLCSM is in place, it would be relatively simple to incorporate new predictive equations when they are available.

Finally, the influence of acidic precipitation on water quality, fish and wildlife is not included in the submodels because, at the time the OLCSM was developed, predictive equations were not available. Research on acidic precipitation is now contributing to a better understanding of these relationships. By incorporating relevant scientific findings in the model, its value could be enhanced for lakes that are acid stressed or sensitive to acidic precipitation.

8.4 SUMMATION

To sum up, the Ontario Lakeshore Capacity Simulation Model is not intended to be immutable. The present version (1982 methods) is capable of predicting the environmental impact of lakeshore cottage development and simulating the effects of various mitigative measures. The next version, including some or all of the recommended refinements, should be tested on lakes with extreme conditions to ensure that it responds to the full range of conditions in the Study Area. Subsequent versions of the OLCSM should be up-dated as required to reflect future advances in the scientific understanding of lake ecosystems.

GLOSSARY

The terms and symbols are defined in the way they are used in this report.

Actions
Lake planning/management strategies.

Algorithm
A rule or series of steps for obtaining a solution to a problem, e.g. Add four numbers ...

Angler-hour
A unit of measurement of fishing pressure describing one angler spending one hour fishing.

Anthropogenic
Generated from human activities.

Areal Hypolimnetic Oxygen Deficit (AHOD)
The oxygen depletion rate in the hypolimnion per unit area, per unit of time, e.g., $mg \cdot m^{-2} \cdot day^{-1}$.

Baseline Condition
A term used to describe the state of a system (e.g. a lake) before any hypothetical changes are made in the lake parameters.

Biomass
Mass of living matter in a given space, e.g. fish biomass in a lake ($kg \cdot ha^{-1}$), or algae biomass per unit volume of water ($mg \cdot m^{-3}$).

Carrying Capacity (K)
The maximum number of organisms (e.g. fish) supported in a given habitat (e.g. a lake) under optimum conditions.

Chlorophyll *a* (CHLOR)
The concentration of the pigment chlorophyll *a* in lake water; a measure of the density of the microscopic free-floating algae (phytoplankton) and primary production in the lake.

Cluster Development
Type of development in which cottage lots are located in groups back from the shore with the adjoining lake frontage providing common shore access.

Computer Code
The logical series of instructions that tells the computer to perform prescribed operations.

Computer Run
The complete execution of all operations of a computer program so that the output or result is produced.

Cottage
A single-family residential building located on the lakeshore or in close proximity to a lake.

Cottage-day
A unit of measurement for cottage use: one cottage in use for one day.

Critical Value
Value of a wildlife indicator variable, indicating that the changes in wildlife habitat threaten either the existence of a wildlife species on the lakeshore or the maintenance of self-sustaining wildlife communities on at least some portion of the lakeshore.

Deer Yard
An area of land where deer congregate in the winter.

Deterministic Model
A model that produces unique output for each set of input data, such as a series of regression equations.

Dissolved Oxygen
The concentration of oxygen in lake water, usually measured in $mg \cdot L^{-1}$ of water.

Driving Variable
In computer modelling terminology, a variable that has a major effect on the output from a model run, e.g. user-days.

Ecological Niche
All interrelationships of a species with its environment, e.g. feeding requirements, habitat and territory.

Ecosystem
An interdependent complex of prevailing environmental factors and biological organisms, e.g. tundra, forest and lake-watershed ecosystems.

Enrichment
The increase in the nutrient concentration of a lake, which stimulates the growth of aquatic life, e.g. plankton, plants.

Fishing Effort
The time spent fishing, usually expressed as angler-hours $\cdot ha^{-1}$ of lake surface.

Flushing Rate (ρ)
The number of times per year (or season) that the water in a lake is completely replaced by inflow from upstream or some other source.

Gaming
Simulating hypothetical conditions in order to understand the direction and magnitude of system responses to stress.

Grid-system
The grid-system is formed from the division of the lakeshore into wildlife segments of uniform width. (See Lakeshore Wildlife Segment)

Hypolimnion
The bottom, non-circulating layer of a lake stratified by temperature.

Indicators
Variables which provide a measure of the health of (or change in) the ecosystem under study, when they are monitored over time.

Indicator Species
Selected animal species included in the OLCSM. The model predicts the impact of development on these species.

Instantaneous Fishing Mortality (F)
The proportion of the fishery removed from the lake per unit of time by anglers (calculated over a short interval of time when fish stock remains at a steady level).

Instantaneous Natural Mortality (M)
The proportion of the fishery that dies due to natural causes per unit of time (calculated over a short interval of time when fish stock remains at a steady level).

Iteration
The repetition of a set of tasks. In the OLCSM, it is each new cycle through some or all of the statements in a computer program.

Lake Morphometry
The physical dimensions of a lake, such as depth, length of shoreline, volume of water.

Lakeshore
In the OLCSM, the lakeshore is defined as shoreland up to 50 m inland from the shoreline of a lake.

Lakeshore Wildlife Segment
When the lakeshore is divided into sections of uniform width (ranging from 50 – 200 m) for the compilation of shoreland habitat data in the OLCSM, each section becomes a lakeshore wildlife segment.

Land Use
The type of use of an area, in this case, the lakeshore, e.g. residential, recreational, agricultural or forest land use.

Littoral Zone
The nearshore shallow water zone of the lake, where light penetrates to the bottom.

Littoral Zone Segment
A segment formed when the littoral zone is divided into sections of uniform width (ranging from 50 – 200 m) for the compilation of littoral zone habitat data.

"Looking Outward"
AEAM Workshop exercise in which information from each of the subsystems is identified by the participants as being required by subsystem x in order to predict its behaviour.

Lot
The part of the registered lot located within the 50-m band of shoreland adjacent to the lake, which is important to wildlife. (Not a registered lot!)

Lot Area
The area of a lot, i.e. lot width (m) multiplied by 50 m.

Management Scenario
A set of hypothetical management strategies designed to reduce the impact on the environment.

Model
An abstraction or simplification of a system.

Multi-tier Development
Development with more than one tier of cottage lots.

Multivariate Statistics
Statistical procedures used to analyze a data set composed of more than one independent and one dependent variable.

Official Plan
A local or regional document, approved under the *Planning Act* by the Ontario Minister of Municipal Affairs, containing objectives and policies established primarily to guide the physical development of a municipality (or part of a municipality), region or unorganized area, while recognizing relevant soil, economic and environmental conditions.

Parameter
In modelling terminology, a constant or slowly changing value that describes a part of the system, e.g. number of kg of phosphorus produced per capita per day.

Phosphorus Binding Capacity (PBCAP)
The amount of cottage-generated phosphorus (mg) that can be retained in the septic system tile bed and in the soil between the cottage and the lake.

Phosphorus Leakage Rate (RLEAK)
The proportion of the cottage-generated phosphorus that leaks to the lake in each season.

Phosphorus Load
The amount of phosphorus per unit of lake surface area entering the lake from all sources, including the watershed drainage, precipitation, upstream lakes and cottages.

Phosphorus Settling Velocity
The amount of sedimentation (m) from the water column per unit of time.

Phosphorus Supply
The total amount of phosphorus entering the lake from all sources, including the watershed drainage, precipitation, upstream lakes and cottages.

Regression
Statistical procedure used to construct an equation with which one can predict values of one variable in terms of one or more other variables.

Secchi Depth
A measure of water transparency; the depth (m) at which a Secchi disc, when lowered into the water, becomes invisible.

Simulation
In modelling terminology, the prediction of changes that occur (under a specified set of conditions) in a system over time.

Single-tier Development
Lakeshore cottage development with one row of cottage lots.

Songbird Likelihood
The probability of finding a particular species of songbird on a lakeshore segment.

Spatial Extent
The geographical area for which the model was designed, i.e. the Muskoka-Haliburton area of Ontario.

Spatial Unit
The unit of land for which the model simulates during a computer run, i.e. in the OLCSM, a single lake surrounded by a 50-m deep band of shoreland.

Stochastic Model
Model with algorithms for natural (random) variation that affects output, e.g. amount of precipitation or seasonal temperatures.

Stratification
Layering of deep lakes by temperature gradients formed by rapid warming of surface waters in spring.

Study plot
Site used for sampling vegetation or observing wildlife (or signs of wildlife activity).

Subdivision
The division of a parcel of land into building lots according to a proposed plan which requires approval under the *Planning Act* before the lots can be registered. The term "subdivision" is also used in the text, under certain specified conditions (i.e. with reference to the land use equations), to describe a group of lots with the same travel mode.

Symbols

Σ	Summation
$kg \cdot m^{-2}$	An example of metric notation used throughout the report; the symbol is expressed as kilograms per square metre.

System
A complex of organized connected parts; in the context of this report, the lake-watershed system.

Time Horizon
Time period over which projections are made on the state of the system, e.g. 10 years, 20 years.

Time Step
Interval of time between model iterations, e.g. monthly, yearly.

Trophic Status
The degree of fertility of a lake, usually characterized by the amount of plant production.

User-day
A unit of measurement for cottage use: one cottage in use for one day by one person.

User-defined Constant
Parameter that does not change over time (or changes so slowly that it can be treated as a constant if re-evaluated periodically) and can be easily changed by the user to more accurately describe known conditions on a given lake or to game with the model.

Variable

Continuous:	A variable that can assume any value with defined limits, e.g. length, weight, amount of rainfall.
Discrete:	A variable that can only change by integer (whole number) amounts, e.g. number of cottages on lake, number of fish/ha.

Watershed
Drainage area comprising all land surfaces that drain (directly or through streams) to a lake.

Zooplankton
The predominantly microscopic aquatic animals that drift and float at various depths in water.

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APPENDICES

APPENDIX A. WORKSHOP PARTICIPANTS

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APPENDIX B. MODEL SENSITIVITY ANALYSIS

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Table B.1 Conditions used in the sensitivity analysis simulation runs

Variable Names	CONDITIONS FOR EACH LAKE SIMULATION RUN 1[g]	2	3	4	5	6	7	8	9	10	11	12	13	14
Mean depth (m)	12.5	*	*	25	12.5	25	25.0	12.5	*	*	*	*	*	*
Surface area (ha)	338		820	820	820	338	820	338						
Watershed area (ha)	1208	4800	1208	1208	4800	1208	4800	1208						
AHOD equation ($mg \cdot m^{-2} \cdot d^{-1}$)	Charlton				Cornett-Rigler			Charlton						
Settling velocity (yr^{-1})	7.4							5.4	9.4	12.4	7.4			
Seasonal settling velocity	0,4.7,0.3,2.4							0,3.4,0.22,1.78	0,6,0.38,3.0	0,7.93,0.5,3.97	0,4.7,0.3,2.4			
Development rate (yr^{-1})	0.20													
Spring oxygen ($mg \cdot L^{-1}$)	10.4										7.8	10.4	10.4	10.4
Fall oxygen ($mg \cdot L^{-1}$)	10.0										10.0	7.8	6.8	10.0
IDRAIN[a]	1													
Cottage-day equation	Accessibility													
CPLK[b] ($\mu g \cdot L^{-1}$)	0.										4.5	0.0		
CMIN[c]	0.													
IHEAD[d]	2													
LT[e] growth rate (yr^{-1})	0.33	0.22	0.44	0.33	0.55	0.33								
SMB growth rate (yr^{-1})	0.50	0.40	0.60	0.50	0.70	0.50								
Proportion of MSY to LT	0.25							0.35	0.45	0.55	0.25			
Proportion of MSY to SMB[f]	0.50													
LT catchability coefficient	0.01										0.05	0.10	0.005	0.01
SMB catchability coefficient	0.0034													
Septic system (1-3) %	0., 95., 5.0													
Natural variation	No													Yes

* Where there is no entry, the condition is the same as in the immediately preceding Lake Simulation Run.
a IDRAIN = 1 if sub-basins exist and their surface areas are known; 0 if sub-basin calculation is not used.
b CPLK = measured initial spring P concentration.
c CMIN = proportion of lots not developed.
d IHEAD = is lake a headwater lake? (2 = yes)
e LT = Lake Trout
f SMB = Smallmouth Bass
g Run 1 = baseline conditions
Run 2-14 = altered conditions

MODEL SENSITIVITY ANALYSIS

Sensitivity analysis is performed in order to determine the set of parameters and variables that influence predictions the most. By using a specialized technique, one can determine the level of error in the parameters that can be permitted before the prediction error becomes unacceptable. In this appendix these parameters and variables are identified and their relationships to selected indicators are described. The sensitivity testing was performed by answering three questions:

1. If one of the driving variables, such as the number of cottages, is changed by one unit, how is that reflected in the indicators generated in the responding submodels?
2. If the input variables such as number of cottages (existing), lake depth or lake surface area, are inaccurately measured, will this cause large errors in the indicators?
3. If some of the assumptions built into the model are incorrect, how much change can be expected in the indicators? What are the key assumptions, variables and parameters of the model?

B.1 METHODOLOGY

A widely applied approach to sensitivity analysis is based on perturbation theory (Tomovic and Karplus 1963). Perturbation theory prescribes that each key parameter is varied about its mean value[1], keeping everything but the parameter or group of parameters unchanged. In this way, the sensitivity to the controlled manipulation of the input parameters of the indicator variables can be determined.

In this analysis, one lake was used and new values were assigned (one at a time) to a set of parameters as a surrogate for the use of different lakes with a range of characteristics. The model was then run after each change and the output was examined (Table B.1). The OLCSM is basically a linear model, that is, variables are related directly and respond either in the same direction or inversely to the perturbed variable. The testing, therefore, was confined to the dominant or controlling variables, such as phosphorous concentration and fishing effort, and selected indicators, such as hypolimnetic oxygen deficit and fish biomass.

As the focus of this work is the impact of cottage development, the sensitivity of major submodel indicators to an increase in development by one unit (a cottage) was analyzed, thus establishing "change per unit-of-loading" rates.

[1] The classical approach requires that all parameters be tested individually and in all combinations using a set of sensitivity equations. A sensitivity equation is created by breaking a mathematical relationship down to its basic components. For example, a hypothetical predictive equation in the model might be:

$$YX = AA + 4.5B$$

where $AA = \frac{6000 - C + ALD}{MM3}$

and $B = 7/WX$, $C = 34 + MM1$, $ALD = XX/4$.

Thus the sensitivity equation, using substitution, would be:

$$YX = (6000 - 34 + MM1 + XX/4) / MM3 + 4.5\,(7/WX)$$

B.2 LAND USE (COTTAGE USE)

The sensitivity analysis of the Land Use Submodel was restricted to the Land Use Component predictive equations. As this submodel generates data on the source of the impact, only points 2 and 3 discussed in the introduction apply here.

The land use indicators, cottage-days and user-days, are calculated using two equations in the Land Use Submodel (Section 4.2). These key formulae are multiple regression equations with both dichotomous and continuous variables. Dichotomous variables are used to describe "either-or" situations and take a value of one or zero only. Therefore, any number preceding this type of variable, e.g. 93.452 * TMODE1, is either included if TMODE1 equals 1 or left out of the prediction if the value is zero. Continuous variables are those which take on any numerical value.

In Equation 1 (Table B.2), TMODE1-3 are dichotomous variables such that, if TMODE1 = 0, 93.452 cottage-days are not used in the annual prediction (however, TMODE2 or TMODE3 will be used). Obviously, giving the wrong value to a dichotomous variable can have a very significant impact on the annual cottage use prediction. To illustrate, using Equation 1 and setting MAXPOORX = 0.4 and DISTSUMH = 4.0, if the travel mode is misidentified as TMODE2 instead of TMODE1, the estimate of cottage-days per cottage is 71 days less than with the TMODE1 value (177 days), which represents a 40% difference.

Table B.2 Equations for predicting annual cottage-days, and the associated cumulative coefficient of determination (R^2) for the variables in the two equations. Equations and data taken from Downing (1986).

ACCESSIBILITY EQUATION (Equation 1)

Annual Cottage-days = 83.105 + 93.452 TMODE1 + 0.322 MAXPOORX − 0.529 DISTSUMH + 22.655 TMODE2 + 21.972 TMODE3

Variable Name	Regression Coefficient	Cumulative R^2
TMODE1	93.452	0.250
MAXPOORX	0.322	0.328
DISTSUMH	−0.529	0.332
TMODE2	22.655	0.337
TMODE3	21.972	0.340

COTTAGER EQUATION (Equation 2)

Annual Cottage-days = 68.312 + 278.578 PERM + 102.042 ESUMMER + 21.291 TMODE1 + 12.623 TMODE2 − 0.191 DISTSUMH

Variable Name	Regression Coefficient	Cumulative* R^2
PERM	278.578	0.838
ESUMMER	102.042	0.911
TMODE1	21.291	0.918
TMODE2	12.623	0.919
DISTSUMH	−0.191	0.920

* R^2 values prior to benchmarking. With benchmarking, the accuracy of the predictions from the two equations is comparable.

The sensitivity of the cottage-day prediction to changes in the independent variables can be determined by examining the regression coefficients and their associated coefficients of determination (R^2). In Equation 1, TMODE1 has a coefficient of 93.452, suggesting that the cottage-day estimate is extremely sensitive to this variable. However, MAXPOORX, with a coefficient of 0.322, has little influence on the final prediction. With regard to R^2 values, TMODE1 alone explains 25% of the variability in the predicted cottage-days while MAXPOORX explains only 7.8% of the variability in the estimate.

In the Land Use Submodel, when using Equation 1 (Table B.2), the predictions are affected most by changes in the TMODE variables, while in Equation 2 (Table B.2) the predictions are influenced most by changes in the cottage typology proportions for PERM and ESUMMER, two continuous variables.

The remaining relationships within the Land Use Submodel, albeit important, did not lend themselves to the sensitivity testing that was conducted.

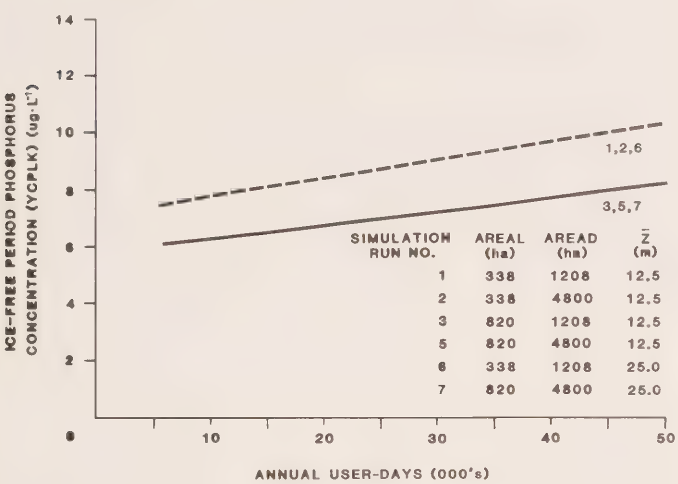

Figure B.1 Relationship of ice-free period phosphorus concentrations (YCPLK) to annual user-days, as lake surface area (AREAL), watershed area (AREAD) and mean depth ($\overline{Z}$) are altered

B.3 TROPHIC STATUS

B.3.1 WATERSHED MORPHOLOGY, COTTAGE USE AND PHOSPHORUS

How sensitive is the Water Quality Submodel to changes in the number of cottages, cottage-days and user-days and what are some of the associated parameters which play a significant role in the phosphorous prediction? As an example, using the actual conditions on the test lake which has a small watershed and a flushing rate of 0.14 yr^{-1}, one cottage unit, receiving 500 user-days of use per year, increased P concentrations by 0.039 $\mu g \cdot L^{-1}$ (Table B.3). The increase is somewhat less on larger lakes due to the greater dilution of the cottage P input (Table B.3).

Phosphorus supplied from sub-basins, i.e. inflow streams, can be calculated by the model in two ways: by using either a global average or a figure based on the sum of all the sub-basin surface areas. Varying the sub-basin area by as much as 75 ha did not affect the phosphorus prediction; in fact, the P values for the baseline and the perturbed scenarios were identical. However, as the error in the measurement of the sub-basin area grows a discrepancy in P values will appear.

Of the three parameters (lake area, watershed area and mean depth) that were changed, ice-free period phosphorous concentrations were most sensitive to variations in lake surface area (Figure B.1). A doubling of the mean depth alone had no influence on P concentrations.

The relationship between units of cottage use and increases in lake nutrients and algal growth will vary between lakes. Soil characteristics and size and age of the development will also influence the amount of nutrients reaching the lake as a result of the soil phosphorus retention capability.

Table B.3 Effect of lake morphometry and watershed size on the relationship between cottage use and lake phosphorus concentration.

Lake Simulation Run	Lake Surface Area (ha)	Watershed Area (ha)	Lake Mean Depth (m)	Predicted User-Days* per Cottage per Year	Annual Increase in Mean Ice-Free Period P ($\mu g \cdot L^{-1}$) per Cottage
1	338	1208	12.5	500	0.039
2	338	4800	12.5	500	0.030
6	338	1208	25.0	500	0.039
3	820	1208	12.5	500	0.018
5	820	4800	12.5	500	0.017
7	820	4800	25.0	500	0.016

* Prediction is the average of all cottages on this lake. User-days per cottage in the Study Area vary from approximately 100-800 user-days per year (Downing, pers. comm. 1983). The global average for the Land Use Component study lakes is 356 annual user-days per cottage.

B.3.2 COTTAGES, CHLOROPHYLL *a* AND SECCHI

In the test lake, with a small surface area and watershed area, one cottage with 500 user-days per year (range in Study Area is 100-800) raised the chlorophyll *a* level by 0.01 $\mu g \cdot L^{-1}$. The combined approximate doubling of the lake surface area and quadrupling of the watershed area (Run #5) indicated a much reduced rise in the chlorophyll *a* concentration in response to an increase in user-days (Figure B.2).

Secchi disc values are derived using the formulation provided by Scheider (1979) where

$$\text{SECCHI} = 5.21/\text{CHLOR}^{0.41} \quad \text{(B.1)}$$

where SECCHI = Secchi depth (m); and
CHLOR = chlorophyll *a* concentration (mg^{-1})

The sensitivity of Secchi to changes in the number of cottages is dampened by at least two filters: the phosphorous – chlorophyll *a* relationship and the chlorophyll *a* – Secchi relationship. Not surprisingly, a substantial change in the number of cottages is required to produce a significant change in the Secchi depth. In the test lake with baseline conditions (Run #1), an increase of 86 lakeshore cottages (between the start and end of the simulation) reduced the Secchi depth by 0.8 m. (Table B.4).

B.3.3 SEDIMENTATION OR SETTLING VELOCITY

To determine the lake phosphorus concentration for any given period, phosphorus losses and gains must be accounted for. A certain amount of phosphorus settles out of the lake, contributing to a reduction in the water-column phosphorus level. *How sensitive is the phosphorus prediction to changes in the rate of phosphorus sedimentation or the settling velocity?* As the model requires both annual and four seasonal settling velocities, both levels of resolution must be examined.

The model is extremely sensitive to changes in the annual sedimentation rate (V). As the rate rises, more phosphorus settles out of the water column (Table B.5). An underestimate of the settling velocity results in a proportional rise in the phosphorus in the water column. However, an overestimate will not elicit an equally dramatic drop in the P concentration (Figure B.3).

Changes in the seasonal allocation of the annual sedimentation rate have little effect on the predictions. Higher spring and lower summer values only slightly increase the final P prediction. Furthermore, a reduction in the summer (May 16 – October 31) and a rise in winter (December 1 – March 15) sedimentation rates tend to cancel each other, yielding an only slightly elevated P concentration after the first year of the simulation.

In summary, the predictions of lake phosphorus concentration are highly sensitive to changes in annual sedimentation rates, as defined by the settling velocity. Changes, up or down, elicit inversely proportionate shifts in the predicted P values. Allocation of the annual values among the four seasons seems to have only a minor effect.

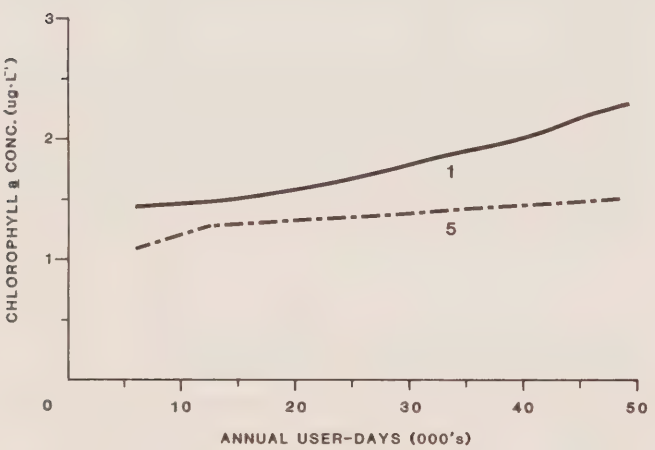

Figure B.2 Sensitivity of the chlorophyll *a* to user-days relationship, to an increase in watershed and surface area (Run #5)

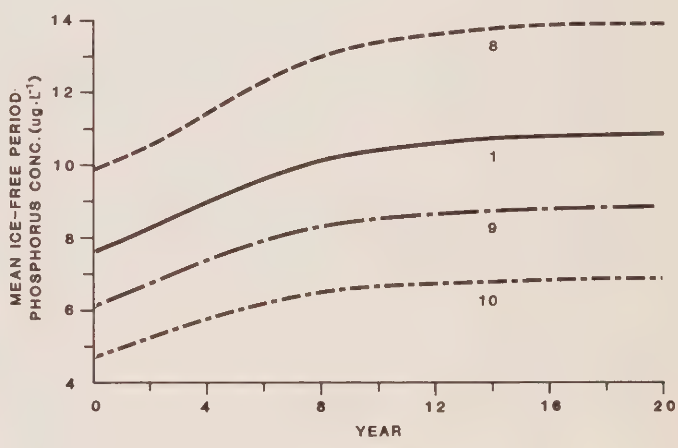

Figure B.3 Mean ice-free period phosphorus concentration as a function of time with different simulated settling velocities (Runs #8, 9 and 10)

Table B.4 Predicted Secchi and chlorophyll *a* values for four simulations. Cottages at start = 12; after 20 years = 98.

Lake Simulation Run	Lake Surface Area (ha)	Watershed Area (ha)	Lake Mean Depth (m)	Secchi Depth (m) Start	Secchi Depth (m) After 20 yrs	Chlorophyll *a* ($\mu g \cdot L^{-1}$) Start	Chlorophyll *a* ($\mu g \cdot L^{-1}$) After 20 yrs
1	338	1208	12.5	4.5	3.7	1.43	2.27
2	338	4800	12.5	4.5	3.9	1.39	1.99
5	820	4800	12.5	5.0	4.4	1.09	1.50
7	820	4800	25.0	5.0	4.4	1.09	1.50

B.3.4 PHOSPHORUS RETENTION AND LEAKAGE RATES IN SOIL

With the existing model, P retention in the soil and the rate at which it leaks (irrespective of retention capacity) from the septic system into the receiving water can be varied. The two parameters are the soil's phosphorus binding capacity per cottage (in kg) termed CAPPC, and the annual proportion of the total human nutrient addition that leaks to the receiving water, termed RLEAK. As RLEAK rises, the maximum amount of phosphorus that can possibly accumulate in the soil drops (Figure B.4). Therefore, *how sensitive is the phosphorus prediction to changes in these two values?* The baseline conditions permit an annual leakage rate of 100% and a soil P adsorption capacity of 2.4 kg per cottage. In the test lake, after 20 years, the phosphorus prediction with baseline conditions rose from 7.6 to 10.8 $\mu g \cdot L^{-1}$ (Run #1, Table B.6 and Figure B.4). If the soil adsorptive capacity is approximately halved and the leakage rate is reduced to 0.2 (Run #7, Table B.6), the phosphorus concentration after 20 years drops by 0.2 $\mu g \cdot L^{-1}$ from the baseline simulation. An increase in the adsorptive capacity to 24.0 kg per cottage (Run #8, Table B.6), a ten-fold increase over the baseline conditions, reduces the lake phosphorus concentration by about 0.5 $\mu g \cdot L^{-1}$ from the baseline conditions. Lowering the leakage rate to 0.01 (Run #9, Table B.6) results in a drop of 1.9 $\mu g \cdot L^{-1}$ from the baseline P level.

These simulation runs suggest that changing the soil's phosphorus binding capacity per cottage has very little effect on the concentration at year 20, whereas lowering the leakage rate clearly reduces the concentration after 20 years. Actually this change is quite small, given an almost 100% reduction in the leakage rate. Neither of these factors, in their present form in the model, appear to elicit marked changes in the ice-free period phosphorus concentration.

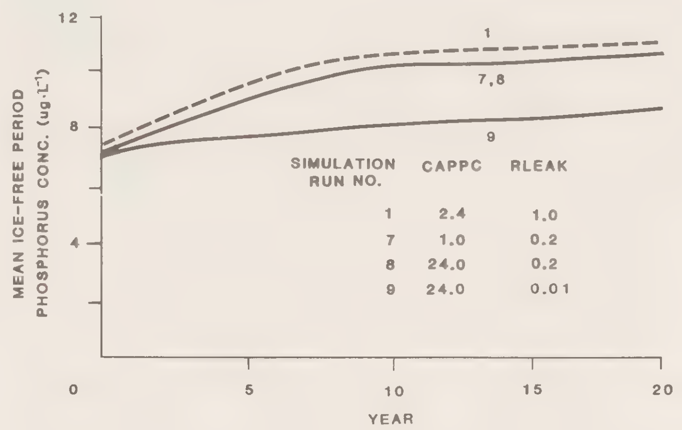

Figure B.4 Mean ice-free period phosphorus concentration as a function of changes in the soil retention capacity (CAPPC) and soil P leakage rate (RLEAK)

Table B.5 Predicted phosphorus concentrations as a function of changing sedimentation rates

Lake Simulation Run*	Lake Surface Area (ha)	Sedimentation Rates (V) Annual	Seasonal Sp	Su	F	W	Changes in Seasonal Distribution of V Relative to Actual	Simulation Results: Mean Ice-free Period Phosphorus Concentration ($\mu g \cdot L^{-1}$) Start	After 20 years	Number of Cottages Start	After 20 years
1	338	7.4	0	4.7	0.3	2.4	Actual values	7.6	10.8	12	98
8	338	5.4	0	3.4	0.2	1.8	Reduced annual V; same seasonal proportion	9.9	13.9	12	98
9	338	9.4	0	6.0	0.4	3.0	Higher annual V; same seasonal proportion	6.1	8.8	12	98
10	338	12.4	0	7.9	0.5	4.0	Higher annual V; same seasonal proportion	4.7	6.9	12	98
									After 10 years		After 10 years
H1	69	12.5	0	4.5	2.0	6.0	Actual values	5.4	14.3	0	78
H2	69	8.0	0	3.0	0.5	4.5	Reduced annual V;	5.8	18.9	0	78
H3	69	18.5	1.5	6.0	3.5	7.5	Higher annual V; same seasonal proportion	4.8	11.4	0	78
H4	69	12.0	1.0	3.0	2.0	6.0	Higher in spring; lower in summer	5.7	14.9	0	78
H5	69	12.5	0	3.0	2.0	7.5	Lower in summer; higher in winter	5.7	14.2	0	78

* 1,8,9 and 10 refer to simulation runs for the test lake (Table B.1); H1-H5 refer to a second test lake.

Table B.6 Parameter values used in analysis of the sensitivity of phosphorus and changes in CAPPC and RLEAK. Existing soil phosphorus binding capacity PBCAP(kg) = 235.2.

Lake Simulation Run	Total Number of Cottages at End of Simulation	Annual Phosphorus Contribution per Person PPC (kg)	Maximum Soil P Storage per Cottage CAPPC (kg)	Fraction of P Bound in Soil that Leaks to Lake RLEAK	Mean Ice-Free Period Phosphorus YCPLK ($\mu g \cdot L^{-1}$) Start	After 20 years
1*	98	0.8	2.4	1.0	7.6	10.8
7	98	0.8	1.0	0.2	7.5	10.6
8	98	0.8	24.0	0.2	7.5	10.3
9	98	0.8	24.0	0.01	7.4	8.9

* Baseline conditions

The timing of the phosphorus release to the lake, assuming a leakage rate of 100%, revealed only slight variations in the P values after 10 years of simulation.

Three scenarios were tested against the baseline conditions (Table B.7):

a) P release uniform over all seasons;
b) P release only in the spring (via sewage lagoon); and
c) P release only in the winter (via sewage lagoon).

The resulting phosphorus predictions, when compared with the baseline values, varied by a maximum of only ± 0.5 $\mu g \cdot L^{-1}$, suggesting that changing the phosphorus release period did little to alter the final (year 10) concentration.

In conclusion, the model predictions of ice-free period phosphorus concentrations on the test lake are not sensitive to changes in seasonal (proportional) phosphorus release.

B.3.5 PHOSPHORUS AND HYPOLIMNETIC OXYGEN

The increase in the lake phosphorus concentration stimulates the growth of aquatic plants and animals. These organisms eventually die and their decomposition consumes oxygen. Decomposition takes place primarily at the lake bottom-water interface, which is the first location to become anoxic, i.e. devoid of dissolved oxygen. To make matters worse, during the major growth and die-off period, lake waters become thermally stratified, restricting the movement and exchange of water between the top layer and the hypolimnion. Therefore, hypolimnetic oxygen concentrations remain low or drop due to a lack of mixing with the oxygen-rich upper layers. As trout prefer the lower, colder region of the lake in summer and as they are quite sensitive to reduced oxygen levels, the model has been programmed to forecast changes in hypolimnetic oxygen concentration. These changes are responsive to fluctuations in the level of nutrient enrichment of the lake.

How sensitive is the hypolimnetic minimum oxygen concentration to variations in P levels as a function of changes in the number of annual user-days?

Table B.7 Predicted phosphorus, minimum summer oxygen concentrations and winter oxygen concentrations under four different simulated annual effluent release schemes*

Effluent Release Timing	Number of Cottages after 10 Years of Simulation	YCPLK Start ($\mu g \cdot L^{-1}$)	YCPLK After 10 years ($\mu g \cdot L^{-1}$)	SSPC Start ($\mu g \cdot L^{-1}$)	SSPC After 10 years ($\mu g \cdot L^{-1}$)	Summer Oxygen Min. Start ($mg \cdot L^{-1}$)	Summer Oxygen Min. After 10 years ($mg \cdot L^{-1}$)	Winter Oxygen Min. Start ($mg \cdot L^{-1}$)	Winter Oxygen Min. After 10 years ($mg \cdot L^{-1}$)
Baseline (seasonal)	90	7.8	10.7	7.7	10.7	7.4	6.6	7.2	7.0
IEVEN=1 steady release year-round	90	7.5	10.2	7.8	10.4	7.4	6.5	7.1	7.0
Lagoon system, all P released in spring	90	7.6	10.9	7.8	11.0	7.2	6.9	7.5	6.3
Lagoon system, all P released in winter	90	7.6	11.0	7.8	11.0	7.2	6.9	7.5	6.3

* Based on additional simulation runs on the test lake, not shown in Table B.1.

YCPLK = mean ice-free period phosphorus concentration
SSPC = mean weighted annual steady state phosphorus concentration
IEVEN = switch to even out cottage use, e.g. as for permanent residences

B.3.5.1 SUMMER

Of the two methods available in the model, the Charlton equation seems slightly more sensitive to variations in P than the Cornett-Rigler formulation. After a 20-year simulation on the test lake, the hypolimnetic oxygen concentration predicted with the Charlton equation dropped by 1.2 mg · L^{-1} with an addition of more than 3 µg · L^{-1} of phosphorus to the system (Figure B.5, Run #1). This rise in P is the result of an 8-fold increase in the user-days from the start of the simulation (Table B.8).

What may account for the different levels of sensitivity for the two equations is that the Charlton formulation uses the chlorophyll *a* value as a key measure of enrichment, whereas the Cornett-Rigler formula uses the lake phosphorus retention as its key variable. Chlorophyll *a* is a more direct measure of the algae production — hence the subsequent plankton die-off and oxidation are more clearly reflected in the AHOD predictions.

The Cornett-Rigler formulation actually predicts a slight improvement in the oxygen conditions (Figure B.5, Run #5). This is probably due, in part, to an overestimate in V, leading to reduced primary production and dissolved oxygen loss.

Thickness and temperature of the hypolimnion

Thickness (ZHYPO) and temperature (THYPO) of the hypolimnion are used in both equations. Cornett and Rigler give more weight to ZHYPO, while Charlton places greater emphasis on THYPO (Chapter 5). If these two variables are perturbed in unison, any significant change would be cancelled out.

Partial "turn-over" assumption

The present model uses the assumption that complete mixing or "turnover" of the water mass will occur every fall and spring. In reality, this may not be true, in which case, instead of starting with an oxygen level of 10.5 mg · L^{-1} in the spring and fall, the biota would have to survive with seasonal maxima of 7 or 8 mg · L^{-1}. This has profound implications for lake trout production, as values below 5 mg · L^{-1} markedly reduce trout survival.

In Run #11, a simulated partial spring turnover resulted in a severely reduced oxygen level by the second year of the simulation (Figure B.6).

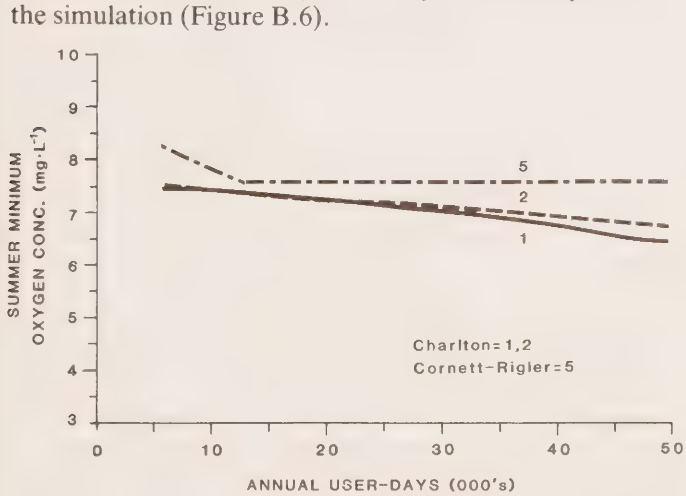

Figure B.5 Comparison of minimum summer hypolimnetic oxygen (SOMIN) predictions for the Cornett-Rigler (Run #5) and the Charlton (Runs #1 and #2) equations

Table B.8 Parameter values and indicator variable results for hypolimnetic oxygen conditions during lake simulation runs 1 and 6

Variable	Lake Simulation Run	TIME Start	TIME After 20 years
Minimum Summer Hypolimnetic Oxygen Concentration SOMIN (mg · L^{-1})	1*	7.45	6.45
	6**	7.60	7.65
Minimum Winter Hypolimnetic Oxygen Concentration WOMIN (mg · L^{-1})	1	7.18	6.93
	6	7.49	7.37
Number of Cottages		12	98
Number of User-days (yr^{-1})		5994	48958
Ice-free Period Phosphorus Concentration YCPLK (mg · L^{-1})	1	7.62	10.83
	6	7.62	10.83
Summer Average Chlorophyll *a* Concentration CHLOR (mg · L^{-1})	1	1.43	2.27
	6	1.43	2.27

* Baseline scenario using the Charlton (1979) AHOD equation
** Baseline scenario using the Cornett-Rigler (1979) AHOD equation with lake depth doubled

In summary, these results suggest that lakes with only partial turnover patterns are significantly more sensitive to development than lakes that are fully dimictic, i.e. experience complete mixing in spring and fall. The nutrient assimilation capacity is severely reduced in lakes with partial mixing, since dangerously low hypolimnetic oxygen conditions develop far sooner under these physical constraints than in lakes with complete mixing.

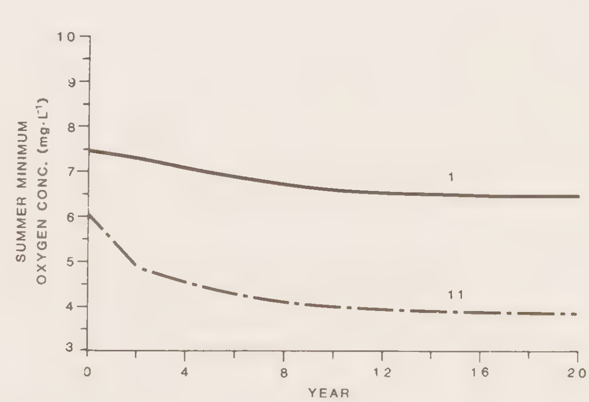

Figure B.6 Comparison of minimum summer hypolimnetic oxygen (SOMIN) curves for a lake with a complete (Run #1) and simulated partial (Run #11) spring water-mass mixing regime

B.3.5.2 WINTER

The minimum oxygen concentration[3] during the winter is calculated using the multiple regression equation of Welch et al (1976), which relies on Secchi depth, mean depth, maximum lake depth and lake flushing rate. Since there is no stratification during the winter, oxygen concentrations are assumed to be homogeneous throughout the water mass. As with the summer season estimate, the assumption is that a complete fall turnover takes place, rejuvenating any anoxic region of the hypolimnion. A 75% fall turnover can result in a severe winter oxygen depression (Figure B.7).

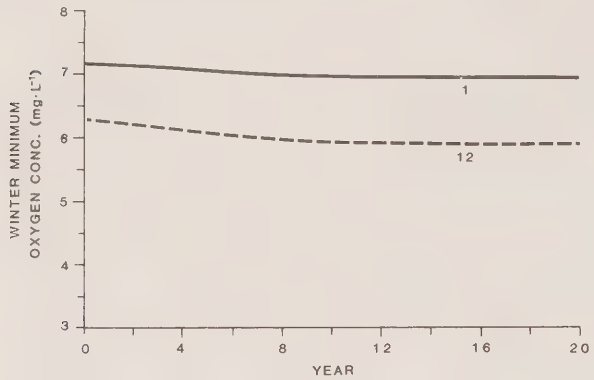

Figure B.7 Comparison of minimum winter hypolimnetic oxygen (WOMIN) curves for a lake with a complete (Run #1) and simulated partial (Run #12) fall water-mass mixing regime

B.3.5.3 IMPLICATIONS

When a partial spring and fall turnover was simulated for a lake with a moderate amount of development (as on the test lake), oxygen levels declined rapidly. Within two years they reached critically low concentrations (<3 $mg \cdot L^{-1}$) for trout and char, fish that cannot survive for long in water with oxygen levels less than 5 $mg \cdot L^{-1}$. Without adequate mixing, anoxic conditions can develop, resulting in "winter kills" once an ice cover has formed on the lake. Mathias and Barica (1980) showed that there is an oxygen concentration threshhold at 4.0 $mg \cdot L^{-1}$ for ice covered lakes, below which any mixing or agitation of the water mass only hastens the oxygen depletion process. This condition is aggravated in shallow lakes.

The oxygen algorithm used in the Water Quality Submodel is most sensitive to the spring and fall oxygen concentrations, which form the starting points for the summer and winter oxygen cycle. Developing the methodology for predicting the amount of turnover is essential for improving the model's ability to project minimum oxygen concentration.

[3] Both winter and summer oxygen equations produce estimates of daily oxygen depletion rates (AHOD) which are then converted to seasonal depletion rates by the following formula: Oxygen depletion in $mg \cdot L^{-1} \cdot season^{-1}$ = (AHOD/ZHYPO * 30/1000) * SEAL(2) where ZHYPO = mean thickness of the hypolimnion (m); AHOD is $mg \cdot m^{-2} \cdot day^{-1}$; SEAL (2) = summer season length in months, i.e. 5.5; 30 = average number of days/month; and 1000 = conversion to volumetric measure. For winter oxygen calculations SEAL (4), i.e. 4.5 months, is used and ZHYPO is replaced by mean depth of lake. The seasonal oxygen depletion rate is then subtracted from the oxygen value at the start of the season, i.e. at turnover, to yield the minimum oxygen level in that season.

B.4 FISHERIES

In the Fisheries Submodel, three elements control the year-to-year fluctuation in population size.

a) physical and chemical condition of the water body, e.g. depth, nutrient concentrations;
b) biological survival factors, i.e. natural growth, birth and death, and
c) fishing activity and fish harvest.

How sensitive is the Fisheries Submodel to changes in the key parameters associated with these three elements?

B.4.1 LAKE MORPHOMETRY AND TROUT PRODUCTION

The basis for the Fisheries Submodel is the morphoedaphic index (MEI), a ratio of total dissolved solids to mean depth. In addition, since one of the variables in the fishing effort estimator is lake surface area, fisheries predictions are generally quite sensitive to changes in the physical measurements of a lake.

B.4.1.1 LAKE TROUT

With a doubling of the lake depth, carrying capacity and trout biomass are initially slightly lower (Figure B.8). The lake productivity is reduced because a smaller proportion of the lake volume is penetrated by light. The biomass at maximum use is similar for depths of 12.5m and 25m. However, an increase in lake surface area by 2.5 times dramatically changes the slope of the line. With an increase in lake surface area, the initial biomass is lower and the decline is considerably more gradual. This suggests that larger lakes have a greater production capacity and, therefore, the fishery in the larger lake will show a more gradual negative response to increased angling pressure than will the fishery in the smaller lake. Increasing simultaneously the depth and surface area depresses the line slightly but does not change its slope significantly.

B.4.1.2 SMALLMOUTH BASS

The relationships described for lake trout also hold true for the smallmouth bass.

B.4.2 NUTRIENT ENRICHMENT AND FISH PRODUCTION

B.4.2.1 LAKE TROUT

Because the total dissolved solids (TDS) are commonly 10^4 times that of the phosphorus concentration, fluctuations in phosphorus have little effect on the TDS (Ryder et al 1974). For this reason the MEI, which is derived from the TDS, is not responsive to nutrient enrichment generated from development. Based on research findings regarding fish productivity (Oglesby 1977b, Matuszek 1978, Prepas 1983) and data from 26 Fisheries Component study lakes, a regression equation was developed to relate the mean ice-free period P and the MEI. This equation provides a mechanism for raising the MEI as phosphorus levels increase due to nutrient contributions to the lake from lakeshore cottages. The adjusted MEI (XMEI), in turn, raises the fish yield estimate.

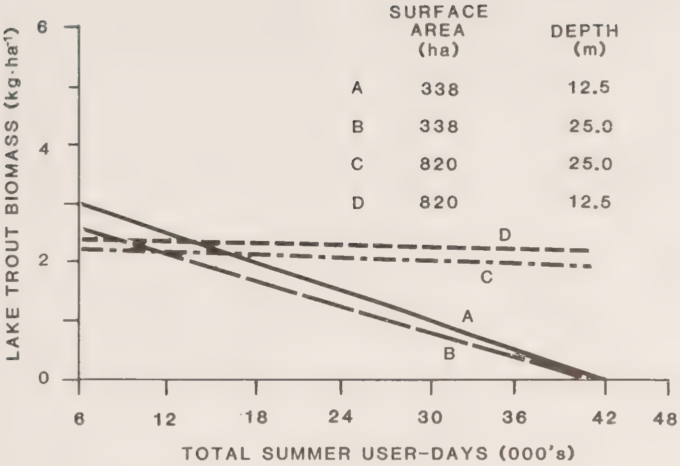

Figure B.8 Influence of changes in lake surface area, lake depth and user-days on lake trout biomass estimates

$$XMEI = 178595.6 * (YCPLK/1000)^{2.3552} \quad (B.2)$$

$r^2 = 0.78$

$n = 26$

where YCPLK = the mean ice-free period phosphorus concentration ($\mu g \cdot L^{-1}$).

By substituting Equation B.2 into Ryder's 1967 MSY formulation

$$MSY = 1.39 * (MEI)^{0.447} \quad (B.3)$$

then

$$XMSY = 1.39 * (XMEI)^{0.447}. \quad (B.4)$$

In highly oligotrophic lakes, nutrient enrichment has a positive influence on lake trout production (represented by carrying capacity K in Figure B.9) by improving the food resources of the fish. However, the opposing actions of enrichment, plant growth-decay and oxygen depletion, can rapidly and severely limit the waterbody's lake trout production capacity by reducing the amount of suitable habitat (K-curve, Figure B.9). In the submodel, this balance is provided by a reduction in the lake trout carrying capacity in response to low oxygen levels.

The enrichment – lake trout production relationship is lake specific. A good knowledge of background enrichment, oxygen concentrations and the lake's water regime is essential for assessing the direction and amount of change resulting from nutrient enrichment of a lake.

B.4.2.2 SMALLMOUTH BASS

In contrast to the response of lake trout, smallmouth bass thrive in enriched waters and only with extreme nutrient additions would the smallmouth bass carrying capacity of the lake (K-curve) start to decline.

B.4.3 NATURAL GROWTH RATE

The relationship between natural fish growth rate and fish biomass (Schaefer 1957) can be shown algebraically:

$$B = K\frac{(1-qE)}{R} \quad (B.5)$$

where
B = fish biomass ($kg \cdot ha^{-1}$);
K = carrying capacity ($kg \cdot ha^{-1}$);
q = fish catchability coefficient;
R = natural fish growth rate; and
E = fishing effort ($\text{angler-hours} \cdot ha^{-1}$).

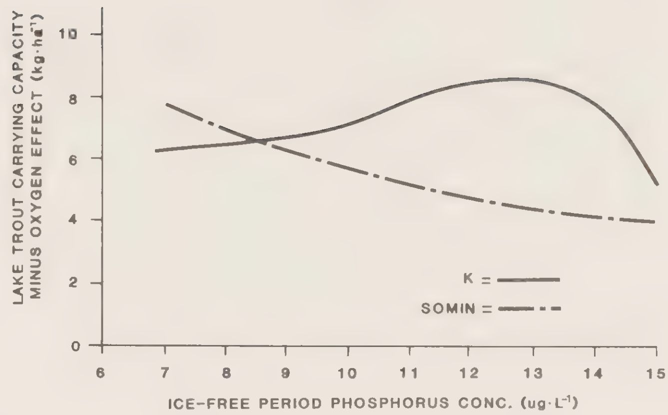

Figure B.9 Sensitivity of lake trout carrying capacity (K) to the drop in the minimum summer hypolimnetic oxygen concentration (SOMIN), as a function of increasing phosphorus (YCPLK)

In Equation B.5, the term in brackets represents the net growth proportion of K. At low natural growth rates (R), initial biomass is very high as a smaller proportion of the biomass has reached recruitable (fishable) size. In this case, many fish are not catchable. The biomass drops off rapidly in response to increasing user-days (Figure B.10). As the growth rate increases, the slope and intercept of the curves approach zero. This pattern occurs because the carrying capacity is inversely proportional to the intrinsic (natural) population growth rate R, that is, as R rises, K and B drop. Using this formulation, high initial fish biomass does not mean that stocks can tolerate intensive fishing. In fact, the opposite is true. With low fishing effort and a poor growth rate (R), initial biomass estimates are high because the stock is underfished (not at a fishable size) and a surplus biomass exists. As the effort rises, coupled with the same slow growth rate, the replacement of harvested fish rapidly loses ground to harvesting (Figure B.10, top curve). With a higher R, there is a higher initial catch and thus a lower initial biomass. However, the biomass declines more gradually due to the faster growth rate and replacement capacity of the population.

In summary, growth rates need to be carefully determined as they can cause markedly different biomass predictions.

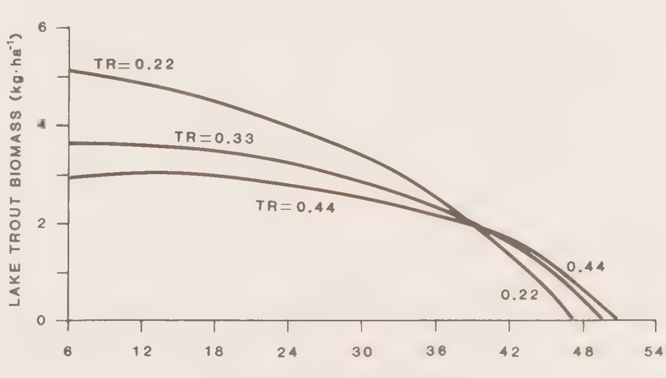

Figure B.10 Sensitivity of biomass predictions to changes in the intrinsic growth rate (TR) for lake trout

B.4.4 CATCHABILITY COEFFICIENT AND TROUT BIOMASS

The catchability coefficient (q) is the proportion of the fish population removed by one unit of fishing effort (one angler-hour). The value of this parameter is difficult to calculate as it varies from species to species, lake to lake, one age group to another and, also, as a function of the fish population density and the skill of the fisherman. The development of q for both lake trout and smallmouth bass for this version of the OLCSM is discussed in Chapter 6. In order to illustrate the model's sensitivity to changes in q values, q was varied from 0.005 to 0.10 and baseline conditions were used for all other input data (Figure B.11).

As q was increased, the biomass dropped off with increasing sharpness in response to a rising number of user-days. The indicators of fish biomass were extremely sensitive to changes in q. Therefore, accurate estimates of q are essential to the production of credible predictions of fish harvest and biomass.

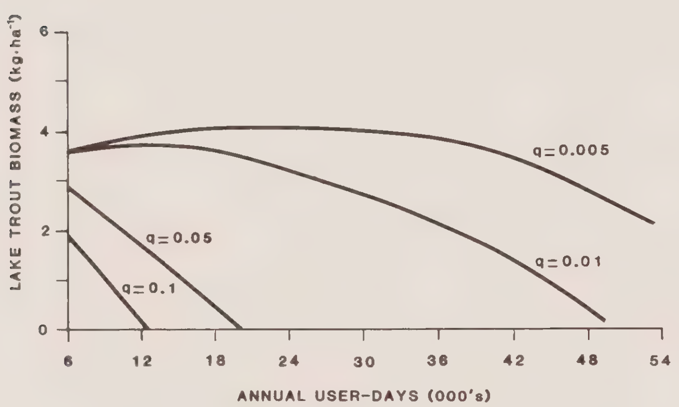

Figure B.11 Sensitivity of biomass estimates for lake trout to variations in the catchability coefficient (q)

B.4.5 FISHING EFFORT FROM COTTAGE-DAYS

Fishing effort is allocated to cottagers and non-cottagers by season. The process is accomplished in two steps: 1) calculate summer fishing effort, and 2) generate winter effort from summer effort through a mathematical relationship.

The submodel program contains two methods for estimating cottager fishing effort in summer. The first method uses the proportion of annual total cottage-days occurring during the summer fishing season, from projections made in the Land Use Submodel; the second method invokes an equation developed by the Fisheries Component researchers (McCombie 1983) which estimates effort from lake surface area and the number of cottages.

The former is discussed here as it is most appropriate to sensitivity testing and uses information supplied directly by the Land Use Submodel. Seasonal use figures are obtained by summing the monthly portions of annual cottage use over a defined number of months. Using this process, a summer use figure (FUSES) is derived which, in turn, is multiplied by four parameters describing fishing habits. The equation is:

$$EFFS = (FUSES * PFISH * ANGLER * HOURS * PTIME) / AREAL \qquad (B.6)$$

		Default Values
where EFFS =	cottager angler-hours in summer per ha of lake;	
FUSES =	cottage-days in summer fishing season;	
PFISH =	proportion of cottages with occupants who fished in summer (Downing, pers. comm.);	0.50
ANGLER =	number of anglers per fishing trip (MNR);	2.00
HOURS =	number of hours per fishing trip (MNR);	2.00
PTIME =	proportion of cottage-days, as defined by PFISH * FUSES, when fishing actually takes place; and	0.33
AREAL =	lake surface area in ha.	

Changing the number of cottage-days allocated to the fishing season has a dramatic effect on the effort estimate. For example, by raising the summer's proportion of annual use by 10%, the variable FUSES rises in direct proportion. The four default values for the fishing habit parameters (PFISH, ANGLER, HOURS and PTIME), when multiplied together, equal 0.66; therefore, summer fishing effort is 0.66 of the summer cottage-days. Estimates of fishing effort would be significantly changed if one or a combination of these parameters were altered.

These four parameters (Equation B.6) are lake-specific but failing anything better, the default values can be used as long as an "average" interpretation is applied.

B.4.6 PARTITIONING FISHING EFFORT AND MSY

In order to make species-specific biomass and harvest estimates, surplus production (MSY) and total fishing effort are divided among lake trout, smallmouth bass and other species. Effort is allocated according to creel census results and is lake specific. Clearly, any changes in these allocations would be directly reflected in the catch estimates.

Maximum sustained yield for the overall fish community is allocated in two stages: first, in terms of the true proportions (according to biomass) and, second, in terms of safe (optimum) levels which may be harvested from the lake indefinitely. These safe levels are designated at 0.25 MSY_c for lake trout and 0.50 MSY_c for smallmouth bass (McCombie 1983, MNR 1982). Any modification in these values would directly alter the MSY and the growth-harvest relationship for these fish species. Great care is needed in determining and/or modifying these parameters.

If no empirical data are available, field experts familiar with the lake should be asked to provide their best estimates.

A summary of the fisheries-related simulation runs and the resulting predictions is presented in Table B.9.

Table B.9 Sensitivity of fish biomass to changes in lake morphometry, water quality, fishing and fish population dynamics

Lake Simulation Run	Scenario tested	Simulation time	User-days	Lake surface area (ha)	Lake mean depth (m)	Annual growth rate	Lake Trout catchability coefficient (q)	MSY All fish ($kg \cdot ha^{-1} \cdot yr^{-1}$)	MSY Lake Trout ($kg \cdot ha^{-1} \cdot yr^{-1}$)	MSY Prop'n for Lake Trout	Summer Lakewide Fishing effort for lake trout ($angler\text{-}hrs \cdot ha^{-1}$)	Nutrient P conc. ($\mu g \cdot L^{-1}$)	Lake Trout Carrying capacity ($kg \cdot ha^{-1}$)	Lake Trout Biomass ($kg \cdot ha^{-1}$)	Lake Trout Catch ($kg \cdot ha^{-1}$)
1	Baseline	Start	5995	338	12.5	0.33	0.01	2.16	0.54	0.25	2.36	7.6	6.5	3.6	0.18
		Yr. 20	48958					2.64	0.66		23.20	10.8	8.0	0.15	0.09
6	Double lake depth	Start	5995	338	25.0	0.33	0.01	1.84	0.46	0.25	2.36	7.6	5.5	3.0	0.15
		Yr. 20	48958					2.64	0.66		23.20	10.8	8.0	0.15	0.09
4	Double lake depth and surface area	Start	5995	820	25.0	0.33	0.01	1.56	0.39	0.25	0.91	6.3	4.8	2.72	0.05
		Yr. 20	48958					1.88	0.47		8.80	7.8	5.7	2.41	0.50
	Changes in Fish and Fishing Parameters														
3	Large lake surface area and increased annual growth rate of fish	Start	5995	820	12.5	0.44	0.01	2.12	0.53	0.25	0.91	6.1	4.9	2.9	0.05
		Yr. 20	48958					2.12	0.53		8.80	7.7	4.9	2.7	0.54
5	Same as 3 but growth rate increased further	Start	5995	820	12.5	0.55	0.01	2.16	0.54	0.25	0.91	6.3	3.9	2.4	0.04
		Yr. 20	48958					2.16	0.54		8.80	7.9	3.9	2.5	0.50
7	Same as 4 but more phosphorus available and MSY proportion for trout is up	Start	5995	820	25.0	0.33	0.01	2.40	0.84	0.35	2.36	9.9	10.2	5.7	0.28
		Yr. 20	48958					3.36	1.20		23.20	13.9	14.6	0.27	0.18
8	Same as 1 but reduced P and available prop of trout MSY is up	Start	5995	338	12.5	0.33	0.01	2.13	0.96	0.45	2.36	6.1	11.7	6.48	0.32
		Yr. 20	48958					2.13	0.96		23.20	8.8	11.7	0.16	0.10
13	Reduced catchability for trout	Start	5995	338	12.5	0.33	0.005	2.12	0.53	0.25	2.36	7.6	6.5	3.64	0.14
		Yr. 20	48958					2.64	0.66		23.20	10.8	8.0	0.60	0.27
11	Slightly reduced initial P but increased trout catchability (Partial spring turnover)	Start	5995	338	12.5	0.33	0.05	2.12	0.53	0.25	2.36	5.3	6.5	3.29	0.49
		Yr. 20	48958					2.64	0.66		23.20	10.8	5.0	0.	0.
12	Increased trout catchability	Start	5995	338	12.5	0.33	0.10	2.12	0.53	0.25	2.36	7.6	6.5	2.91	0.87
		Yr. 20	48958					2.64	0.66		23.20	10.8	8.0	0.	0.

B.5 WILDLIFE HABITAT

The Wildlife Submodel contains equations and relationships which provide estimates of the response of birds, small mammals, furbearers, ungulates and herptiles to changes in their natural habitat.

As this submodel deals with over 30 animal species, each with varying sensitivities to habitat removal, a detailed analysis of each species is beyond the scope of this report. However, some generalities are appropriate.

B.5.1 HABITAT DISTURBANCE

Wildlife researchers identified a strong relationship between the lot area (lot width (m) * 50 m) and the proportion of the area disturbed in the ground, shrub and tree vegetation layers. Although the actual area disturbed was independent of lot size, the proportion of the lot disturbed decreased as the lot size increased (Figure B.12).

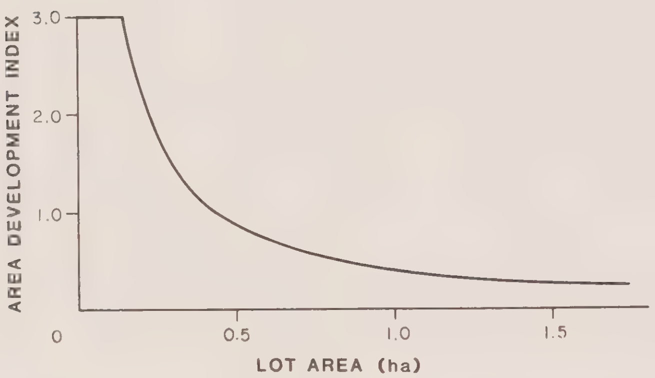

Figure B.12 Area Development Index and its relationship to cottage lot size

B.5.2 SMALL MAMMALS

The seven small mammals studied vary in their sensitivity to development. An overriding sensitivity criterion for all species is their response to the removal of preferred habitat. The sensitivity of these animals to change in their habitat is shown in the complex, sensitivity response, surface diagrams (Figure B.13). For example, the graph for the deer mouse suggests that this species is sensitive to development, as indicated by the steep slope of the graph along the development axis. Moreover, this species' sensitivity to a shift from deciduous to coniferous tree cover at low levels of development is illustrated by the slope of the line from the peak of the graph and along the axis of coniferous composition. Sensitivity to a change in forest cover type is pronounced for the red squirrel. Any removal of coniferous tree cover, as by the selective cutting of coniferous trees, severely reduces this species' usable habitat.

Four of the seven species studied appear to prefer deciduous tree cover and the associated understorey plant community (Racey and Euler 1982). Clearly small mammals are sensitive to changes in the forest cover type.

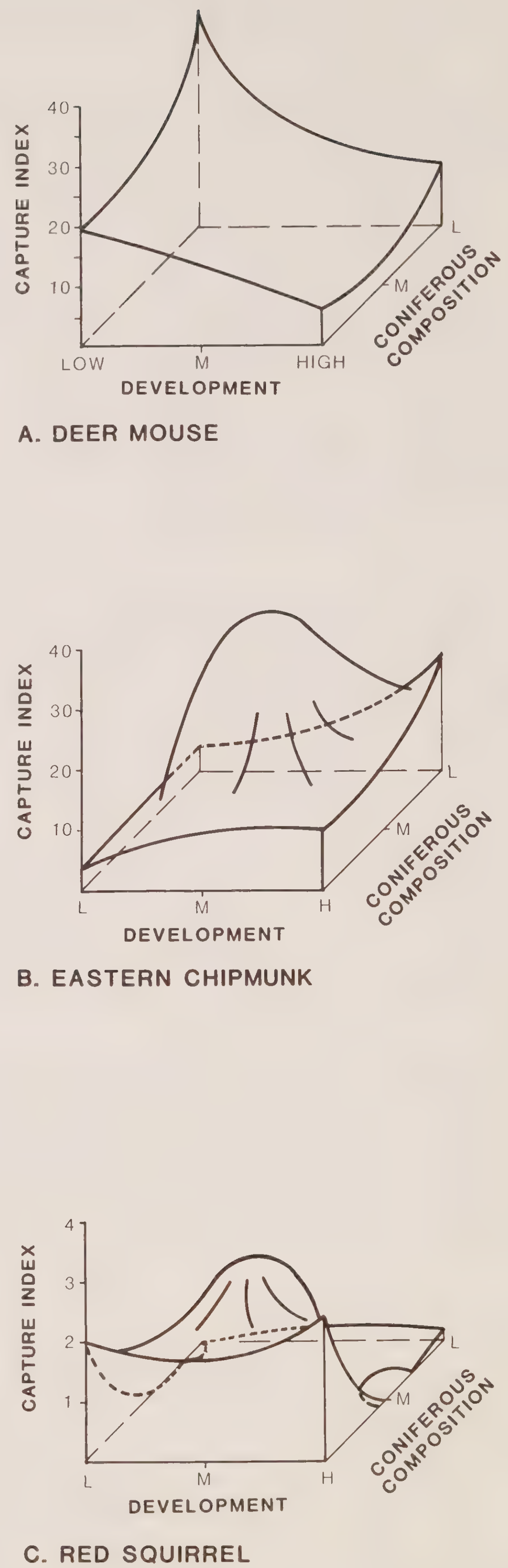

Figure B.13 Response of three small mammal species to changes in the level of development and the forest cover composition (from Euler 1983)

B.5.3 MINK

Mink prefer coniferous habitat and are seriously affected by disturbance from lakeshore development (Figure B.14). The relationship of the Area Development Index (amount of disturbed habitat) to the mink activity index (Figure B.15) suggests that mink populations may decline to 20% of their activity level (population size) with a 0.5 unit increase in the Area Development Index.

As with the small mammals, careful identification of the type of tree cover along the lakeshore is essential for providing sound predictions of the mink's response to development.

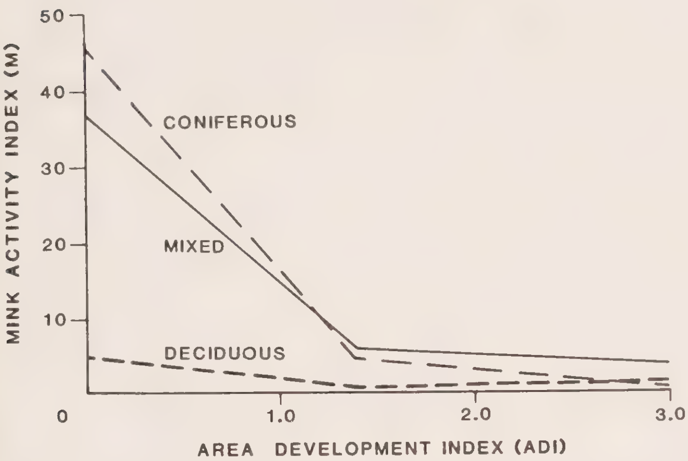

Figure B.14 Mink activity as a function of the Area Development Index, for three forest cover types

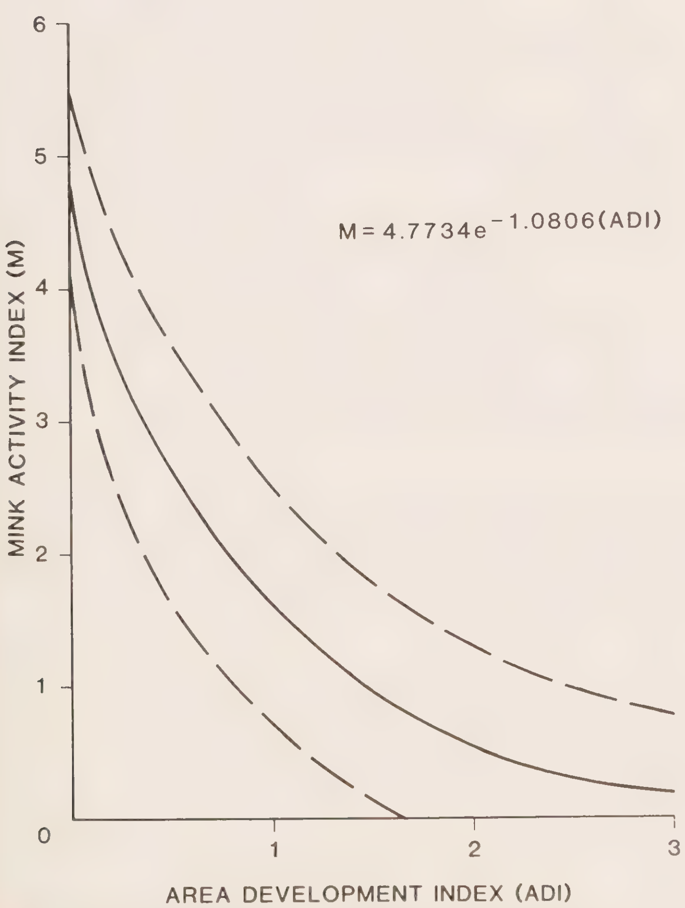

Figure B.15 Relationship of the Mink Activity Index (M) to the Area Development Index (ADI)

B.5.4 SONGBIRDS

Songbirds respond rapidly to a change in the type of tree cover. Species associated with areas of low tree volume, low tree density and little understorey can tolerate habitat disruption, while species associated with dense, closely-spaced stands are intolerant of habitat alteration. The 19 species examined were most sensitive to canopy volume, tree density and amount of understorey. Here, as with the small mammals, it is the change in type of forest cover which determines the degree of response by the song-birds.

Wildlife biologists have established critical threshhold levels of disturbance for each of the songbird species. These are programmed into the submodel and, at each iteration, they are compared with the predicted disturbance to determine the amount of suitable songbird habitat.

B.6 NATURAL VARIATION

The model contains a built-in feature which permits the user to add natural variation to the input parameters (Section 5.3). Presently this is done only in the Water Quality Submodel. However, as output from this submodel is used in the Fisheries Submodel, the variation is transferred to the fisheries indicators. The sensitivity of the water quality and fisheries indicators to natural variation is examined here.

Mean ice-free period phosphorus concentration changes markedly with natural variation (Figure B.16 a & b). After a ten-year simulation on one test lake, the naturally-varied P value was 0.7 $\mu g \cdot L^{-1}$ less than the estimate made without natural variation. A further example (Figure B.16 c & d) shows the total annual phosphorus delivered to the lake (YPLOAD) and the total P flushed downstream (SDOWNP). The phosphorus retained in the lake (shaded area on graph) varies significantly from the run without random variation. There is a certain amount of "compensation", as during wet years the greater input of nutrients from the watershed is countered by increased downstream flushing; the opposite is true for dry years. However, in this example there seems to be greater flushing than input, resulting in a reduced phosphorus load in the lake after 10 years (Figure B.16 d).

Other water quality indicators are affected when natural variation is implemented in the model. Oxygen, however, responds inversely to phosphorus, i.e. as P supplies drop, oxygen concentrations rise.

With natural variation applied (Figure B.16 f) in the Fisheries Submodel, trout biomass levels are as much as 0.5 $kg \cdot ha^{-1}$ below the standard run (Figure B.16 e), resulting in lower surplus fish production.

In conclusion, the application of natural variation suggests that resulting predictions will fluctuate by as much as 30% from the results of non-naturally varied runs. In view of this high level of sensitivity, it is recommended that simulations be run with the natural variation option.

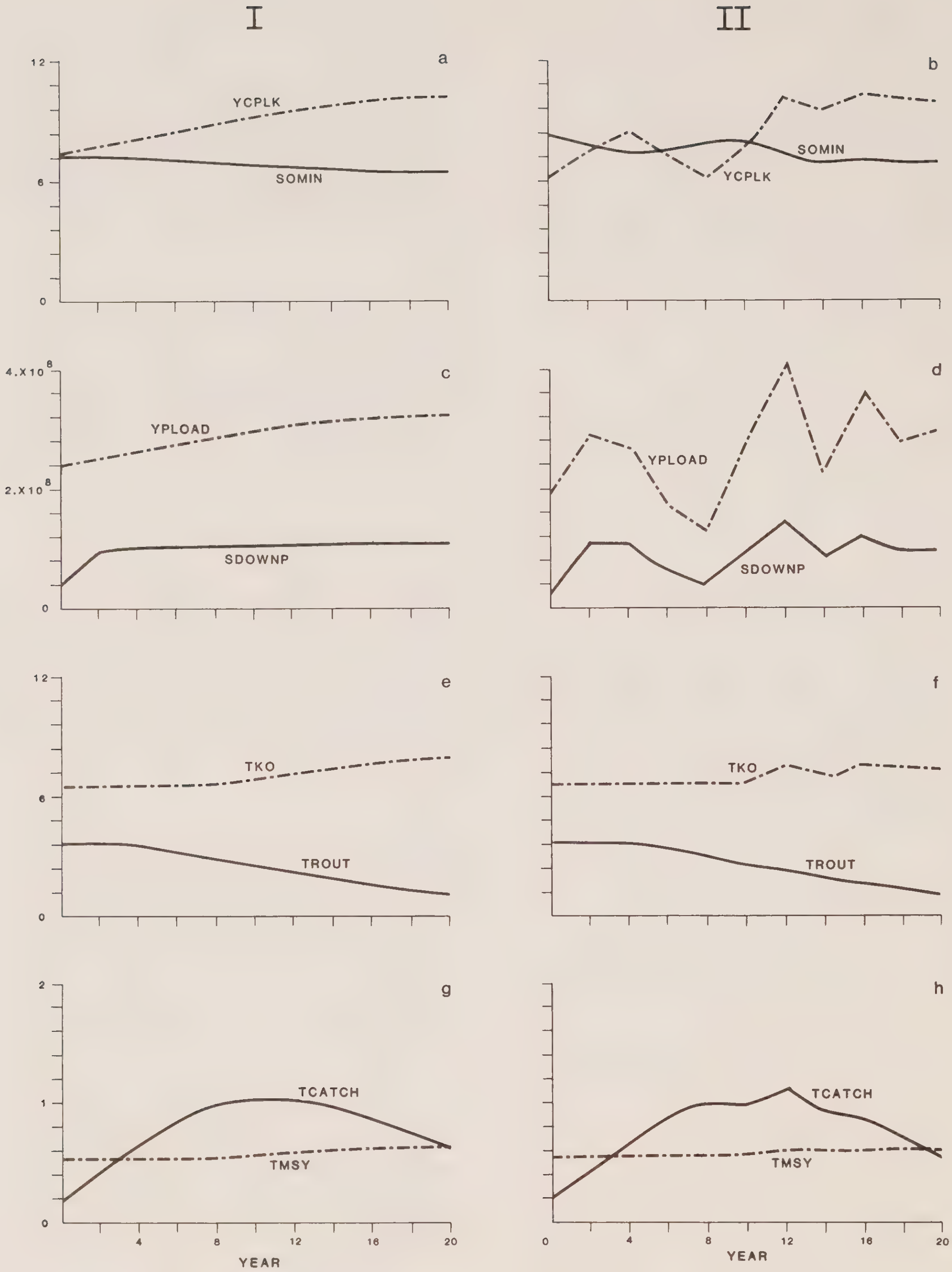

Figure B.16 Comparison of predictions without (I) and with (II) natural variation for mean ice-free period phosphorus concentration (YCPLK) and summer hypolimnetic oxygen (SOMIN); total phosphorus supply (YPLOAD) and total downstream losses of phosphorus (SDOWNP); lake trout carrying capacity (TKO) and biomass (TROUT); and lake trout maximum sustained yield (TMSY) and catch (TCATCH)

APPENDIX C. AN EXAMPLE OF MODEL APPLICATION

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TABLES

FIGURES

AN EXAMPLE OF MODEL APPLICATION

The method of applying the OLCSM and interpreting the model output is demonstrated in the following example of a proposed plan of subdivision on hypothetical Lake Z in the Muskoka-Haliburton area of Ontario.

Lake Z is an oligotrophic lake with a lake surface of 350 ha. The lake supports a lake trout fishery and other sport fish species, including smallmouth bass, lake whitefish and yellow perch. Winter fishing is permitted on the lake. A public boat launching ramp provides boat access for non-cottagers.

Existing lakeshore development consists of 65 cottages and 20 vacant lots. The proposed plan of subdivision includes 13 lots, arranged in a single tier along the lakeshore. If cottages are constructed on all vacant and proposed lots, during the course of the simulation, there will be a 51% increase in the number of cottages. Roads leading to the existing cottages, vacant lots and proposed lots are hard-surfaced and open in winter.

In order to predict cottage-days and user-days, it is essential to know 1) the type of existing access to the cottages and lots; 2) whether there is an official plan for the municipality or region and, if there is, the policies regarding permanent and seasonal use; 3) whether such policies have been implemented through zoning regulations; and 4) whether there are plans for roads that would change the existing type of access to the subdivision, the distance to the nearest high order urban service centre or the distance on poor roads.

In the case of Lake Z, there is neither an official plan nor a zoning by-law containing restrictions on permanent use for the area, so it is assumed that some cottages may be used all year. Further, there are no plans for new roads or improvements that would change the type of access to existing or proposed lots.

C.1 OLCSM PREDICTIONS

During the period of the simulation, cottages are assumed to be constructed on all remaining vacant lots, at a rate of 20% annually. The same rate is applied to the proposed subdivision, which is started in year three. The length of each simulation, in this case, is 20 years — a long-term planning horizon.

The conditions for each scenario appear in Table C.1. The model is run with the existing development (Scenario 1), followed by the addition of a proposed subdivision (Scenario 2). Scenarios 3 through 6 simulate the impact of both existing and proposed development when selected mitigative measures are applied.

Table C.1 Specified conditions for Lake Z simulations

SCENARIO Variables	1 Existing Development	2 Existing & Proposed Development	3 Scenario 2 Plus No Winter Fishing and Holding Tanks	4 Scenario 2 Plus Summar and Winter Fishing Restrictions and Lot Relocation	5 Scenario 2 Plus Redistribution of Fishing Pressure, No Winter Fishing and Increased Lot Width	6 Scenario 2 Plus Road Closed in Winter
Existing number of occupied lots	65	65	65	65	65	65
Existing number of vacant lots	20	20	20	20	20	20
Proposed number of lots	0	13	13	13	13	13
Total number of lots	85	98	98	98	98	98
Sewage disposal system	septic	septic	holding tanks on proposed lots	septic	septic	septic
Summer fishing season	May-Sept.	May-Sept.	May-Sept.	July-Sept.	May-Sept.	May-Sept.
Winter fishing	yes	yes	no	no	no	yes
Proportion of angling effort for lake trout	0.60	0.60	0.60	0.60	0.30	0.60
Proposed subdivision design	NA	unchanged	unchanged	lot relocation	double lot width	unchanged
Winter roads	yes	yes	yes	yes	yes	no

SCENARIO 1: EXISTING DEVELOPMENT

The existing development consists of 65 cottages and 20 vacant lots on which new cottages will be constructed during the simulation.

The model projections indicate that the water quality indicators (Figure C.1 C) in year 1 of the simulation (present conditions) are marginally within the Ministry of the Environment's water quality level #1. This level represents the range of phosphorus (1-10 $\mu g \cdot L^{-1}$) and chlorophyll *a* (0-2 $\mu g \cdot L^{-1}$) concentrations in unenriched waters, which provide optimal habitat conditions for cold water fish species, such as lake trout.

The lake nutrient concentration increases during the simulation as a result of the additional user-days associated with cottages constructed on existing lakeshore lots. After the 20-year simulation (Table C.1, Scenario 1), the water quality indicators reflect levels of nutrient enrichment found in moderately enriched waters, i.e. water quality level #2. This level is acceptable for body contact recreation, such as swimming and water-skiing, as well as boating and fishing. However, level #2 is not compatible with the preservation of the cold water fisheries.

Nutrient enrichment has a potentially negative effect on lake trout, due to oxygen depletion. The minimum oxygen concentrations in the simulation (6.6 $mg \cdot L^{-1}$) approach the 5-6 $mg \cdot L^{-1}$ range, which represents the minimum levels to which lake trout can be exposed for extended periods without adverse effects (Figure C.1 C). Since the summer minimum oxygen prediction reflects the average value in the hypolimnion, a value close to this range may result in dangerously low levels at the sediment surface. This suggests possible stress on the lake trout, through reduction in the amount of suitable habitat.

Overfishing for lake trout is indicated from the estimate of lake trout catch at the start of the simulation. In year one, it is double the maximum sustained yield, or the surplus production, for that species (Figure C.1 D). The nutrient additions which improve the fish food base are not great enough to offset the effects of the existing high level of fishing activity on the lake. The result is a steep decline in the lake trout stock, which causes virtual extinction of the population.

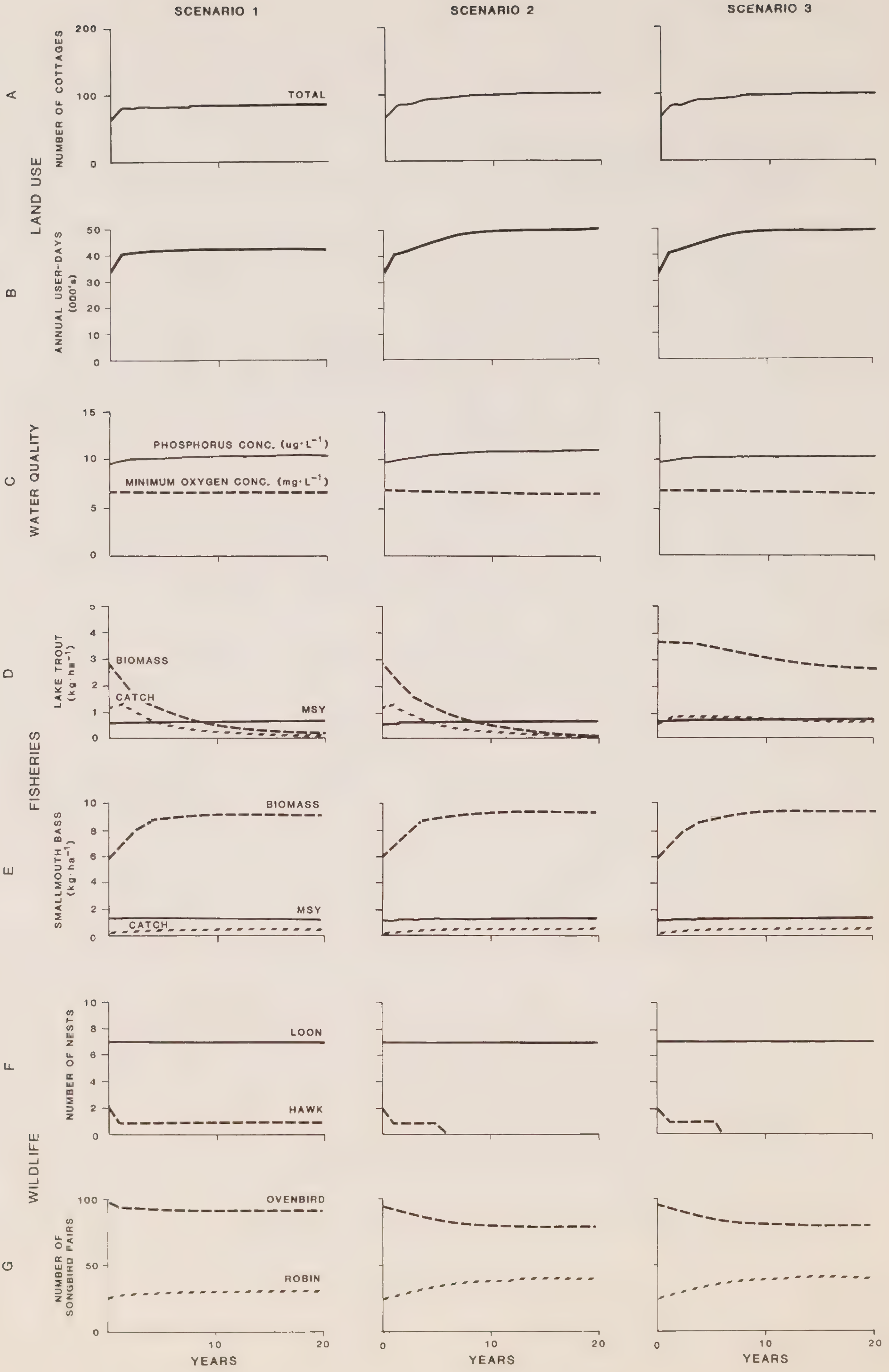
The smallmouth bass catch, on the other hand, remains well below the maximum sustained yield (Figure C.1 E). The smallmouth bass population actually increases for the first five years, then stabilizes at a higher biomass level. This species is less sensitive than lake trout to angling pressure and reduced oxygen concentration. In addition, there is some evidence that rock cribs used as dock foundations, which are associated with cottage development, provide shelter for young bass.

The seven potential loon nesting sites identified on the lake are located on small islands more than 50 m from the shoreline, so it is assumed that these nesting sites are undisturbed by the additional cottages (Figure C.1 F). However, when new cottages are built on existing vacant lots, one of the two potential broad-winged hawk nesting sites, being close to a cottage, will no longer be suitable for that purpose. The preservation of at least two suitable nesting sites for each species (broad-winged hawk and common loon) is recommended by the Ministry of Natural Resources wildlife scientists, in order to ensure the probability of at least one nesting pair of each species on a lake.

The American robin is an example of a songbird species that is tolerant of development. The number of breeding pairs of robins around the lake increases (Figure C.1 G) as cottages are constructed on existing vacant lots, creating more open habitat. On the other hand, songbird species such as the ovenbird cannot tolerate the vegetation changes associated with development, as these species typically inhabit undisturbed forests.

Examination of other lakeshore wildlife indicates low levels of disturbance. For example, more than 70% of the original songbird habitat and mink activity are retained at the end of the simulation.

It is apparent from this simulation that the existing development is exerting stress on the fishery and the lake water quality.


SCENARIO 1
SCENARIO 2
SCENARIO 3
A
B
LAND USE
NUMBER OF COTTAGES
TOTAL
ANNUAL USER-DAYS (000's)
C
WATER QUALITY
PHOSPHORUS CONC. (ug·L⁻¹)
MINIMUM OXYGEN CONC. (mg·L⁻¹)
D
E
FISHERIES
LAKE TROUT (kg·ha⁻¹)
BIOMASS
CATCH
MSY
SMALLMOUTH BASS (kg·ha⁻¹)
BIOMASS
MSY
CATCH
F
G
WILDLIFE
NUMBER OF NESTS
LOON
HAWK
NUMBER OF SONGBIRD PAIRS
OVENBIRD
ROBIN
YEARS
YEARS
YEARS

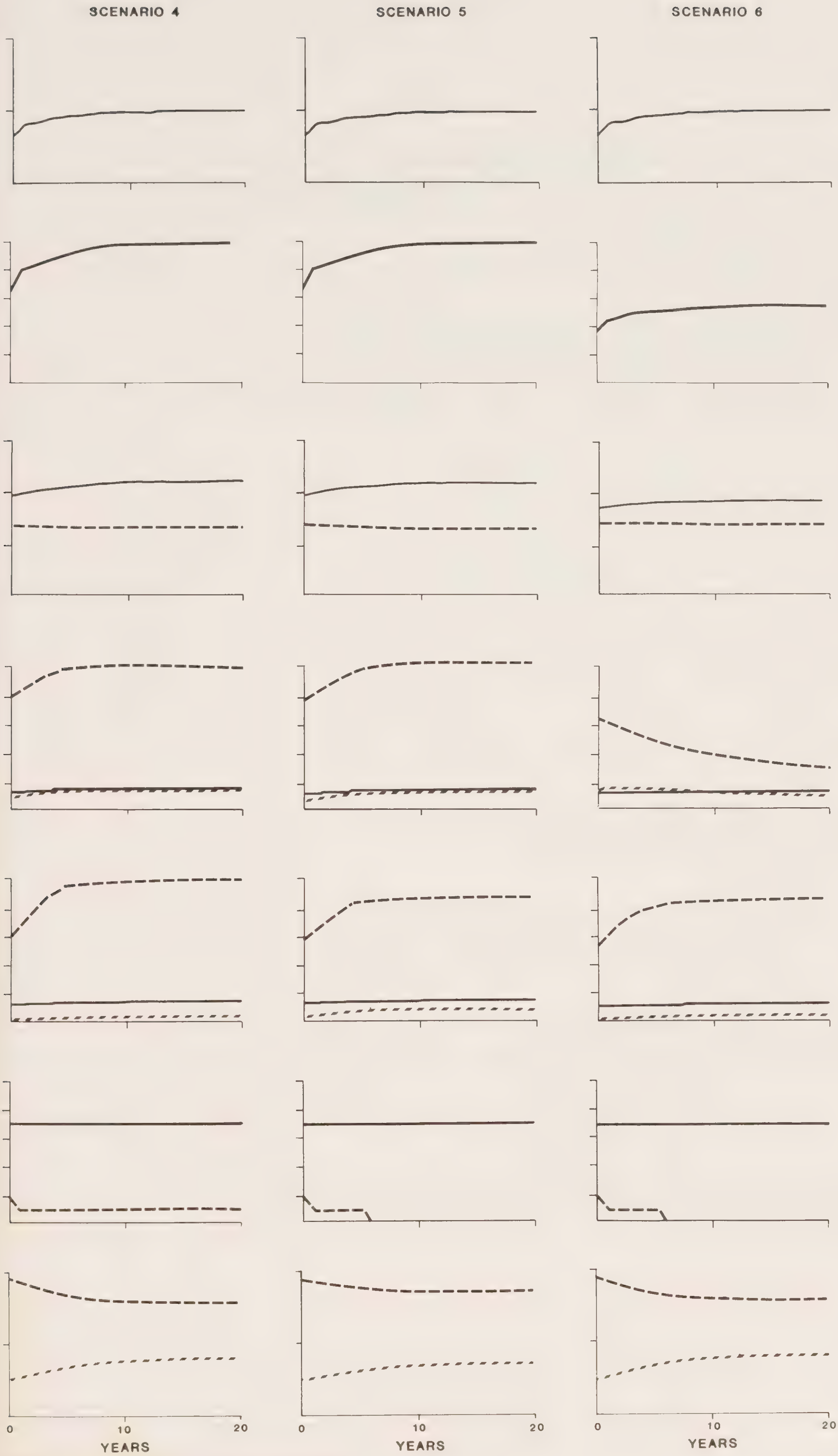


Figure C.1 An example of model application

SCENARIO 2: EXISTING AND PROPOSED DEVELOPMENT

The addition of 13 proposed lots to the existing development increases the total number of lakeshore lots to 98 (Table C.1, Scenario 2). When cottages with septic tanks are constructed on the 13 lots, lake enrichment is increased by a small amount. The 13% increase in the number of annual user-days, however, places the lake marginally within the moderately enriched category of lakes (water quality level #2).

The lake trout population declines slightly faster than with the existing development alone (Scenario 1), due to additional lake trout angling pressure from the proposed cottages. The new anglers increase the high level of fishing activity associated with the existing development.

Although the smallmouth bass angling effort increases, this pressure is offset by additional nutrients in the water, which increase the fish food base. The result is a net increase in the smallmouth bass biomass in the lake.

The number of suitable loon nesting sites is the same as in Scenario 1. However, because of the proximity of the hawk nesting site to the proposed cottages, the one remaining potential nesting site is lost.

As in Scenario 1, the number of robins increases but the addition of the proposed cottages causes a greater reduction in the number of ovenbirds, due to vegetation removal on the cottage lots.

Other wildlife species that are intolerant of development, such as mink, experience slightly more disturbance under Scenario 2, although the wildlife indices remain substantially above the levels designated by wildlife scientists as a minimum for preservation of these species around the lake.

The environmental indicators suggest that management strategies need to be considered for both the existing and proposed development. In order to mitigate the predicted environmental stresses in Scenarios 1 and 2, four additional scenarios were simulated (Table C.1, Scenarios 3 – 6), each involving the introduction of selected management strategies.

SCENARIO 3: EXISTING AND PROPOSED DEVELOPMENT (98 COTTAGES)

□ **Holding tanks in proposed subdivision (13 lots)**

□ **Winter fishing restrictions**

Holding tanks provide 100% removal of domestic sewage from the cottage site by disposal of the effluent in an acceptable manner, usually in a sewage treatment plant. The condition for subdivision approval specifying holding tanks prevents any further enrichment of the lake from human sources.

With the closing of the winter fishery, the lake trout catch at the start of the simulation is below the maximum sustained yield for the species (Figure C.1 D). Following year one, the lake trout summer harvest rises to exceed the maximum sustained yield, due to the increased number of cottages and user-days. This results in a steady lake trout biomass level for the first four years and a gradual decline in the biomass thereafter. Although the decline in the lake trout population is not desirable, over the long run the results may not be catastrophic. There is time for further scrutiny of the situation and re-evaluation of control measures.

The smallmouth bass population is unaffected by these restrictions since smallmouth bass are not a part of the winter harvest. They seek the warmest water at the bottom of the lake in winter and are relatively inactive.

SCENARIO 4: EXISTING AND PROPOSED DEVELOPMENT (98 COTTAGES)

- **Summer and winter fishing season restrictions**
- **Lot relocation in proposed subdivision**

Reduction in the length of the summer fishing season from May through September to July and August, plus the closure of the winter fishery, dramatically reduces the overharvesting of lake trout. The trout harvest remains below the maximum sustained yield throughout the 20-year simulation. The lake trout biomass increases during the first eight years and then stabilizes. The increase in lake trout can be attributed primarily to underfishing, that is, the incomplete harvest of the surplus production, which causes a rise in the total lake trout stock.

Due to the shorter summer fishing season, fishing pressure on the smallmouth bass is reduced. Because fewer bass are removed from the lake, the population rises to higher levels than in Scenario 2.

In year one of the simulation, one of the two potential hawk nesting sites is unsuitable, due to the construction of cottages on existing vacant lots. The potential of the other nest site is also eliminated by its proximity to cottages in the proposed subdivision (Scenario 2). To retain the hawk nesting potential, relocation of the proposed lots could be considered as a condition of subdivision approval. By developing at another location, the nesting site remains a safe distance from the source of disturbance. The number of lots in the proposed subdivision, in this case, is not reduced.

SCENARIO 5: EXISTING AND PROPOSED DEVELOPMENT (98 COTTAGES)

- **Redistribution of fishing pressure**
- **Winter fishing restrictions**
- **Larger lot sizes in proposed subdivision**

On many lakes in the Muskoka-Haliburton area, anglers are over-harvesting the cold water fish species, especially lake trout, and under-harvesting the warm water species. As stated in the MNR Minden District Land Use Strategy (Draft), a redistribution of fishing pressure from trout to non-trout species is necessary to meet the sport fish target set by the District (Ontario Ministry of Natural Resources 1982a). In some cases, lakes could sustain additional fishing pressure, if it were redirected to underutilized species.

In Scenario 5, the conditions of the simulation include a redistribution of angling effort from 60%:35% to 30%:65% (lake trout:smallmouth bass), thereby reducing the effort for lake trout by half and nearly doubling the effort for bass. Five percent of the total fishing effort is allocated to other species. In this case, when combined with restrictions on the winter fishing season, the lake trout catch remains within the estimated surplus yield for that species. Although the total number of angling hours is the same as in Scenario 3, the lake trout biomass no longer indicates a declining population. Where there is less severe overfishing, the redistribution of angler effort alone may stabilize the lake trout population. This strategy could be implemented by introducing fish quotas or undertaking an educational program to encourage voluntary restraint.

Lot width affects the amount of vegetation removed for cottages and, in turn, the impact on wildlife species that cannot tolerate changes in their habitat. By doubling the width of the proposed 30-m lots, more ovenbirds are retained on the lakeshore. The decline in breeding pairs of this species is less pronounced than with smaller lot sizes (Scenario 2). In contrast, species such as robins, that find suitable habitat in or near cottage clearings, do not increase as much under these conditions of reduced cottage density.

SCENARIO 6: EXISTING AND PROPOSED DEVELOPMENT (98 COTTAGES)

□ **Roads closed in winter**

This scenario is simulated to illustrate the implications of cottage accessibility with respect to cottage use. It is assumed that roads are closed in winter. Therefore, although a car can be used for summer transportation to the cottage, some other travel mode will be necessary to complete the trip to the cottage in winter.

Where roads are closed in winter, the amount of cottage use is considerably lower and non-cottager use of the lake in winter is reduced. In this case, 44% fewer user-days result in less nutrient enrichment of the lake and less fishing activity than in Scenario 2.

Even though the number of cottages is the same as in Scenario 2, the closed roads in winter virtually rule out the presence of permanent residents. Because permanent residents occupy their lakeshore cottages 365 days per year (compared with the average cottage use of 116 days per year), a high proportion of this group on a lake increases the total number of annual cottage-days.

The lake trout population declines gradually, due to overfishing, during the first eight years of the simulation. However, the impact may not be catastrophic. More stringent fishing restrictions or, perhaps, redistribution of angling effort might correct this situation.

C.2 OVERVIEW OF SIMULATIONS

The example demonstrates the method of predicting, interpreting and mitigating the impact of cottage development on a hypothetical lake. Other examples could be used to demonstrate that the same number of cottages may have quite a different impact on another lake with different lake and watershed characteristics. In addition, if cottages are built on an upstream lake, they may increase the phosphorus concentration in the lake and in the outflow streams, thus possibly affecting the cottage capacity of the downstream lake.

The effects of variations in such factors as number and location of cottages, type of cottage access, timing of construction, lake and watershed characteristics, type of wildlife habitat and selection of management strategies can all be simulated by applying the OLCSM. The resulting predictions of probable impact can assist planners in guiding development in ways that will avoid damage to the inland lake environment.

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LAKESHORE CAPACITY STUDY

MICROBIOLOGY

LAKESHORE CAPACITY STUDY

MICROBIOLOGY

MARCH 1983

Prepared by:
C.A. BURGER B.Sc.
Ministry of the Environment

For:
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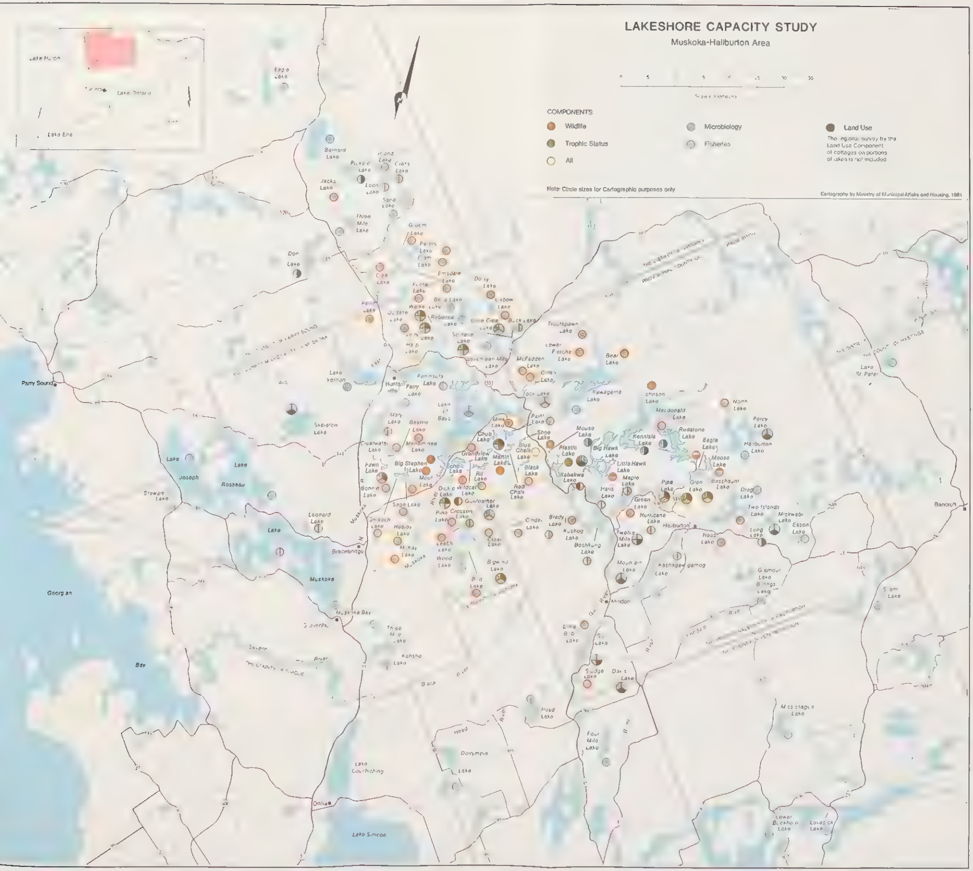
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Muskoka-Haliburton Area
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Wildlife
Trophic Status
All
Microbiology
Fisheries
Land Use
The regional survey by the Land Use Component of cottages on portions of lakes is not included
Note: Circle sizes for Cartographic purposes only
Cartography by Ministry of Municipal Affairs and Housing, 1981
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Lake Ontario
Lake Erie
Parry Sound
Huntsville
Bracebridge
Muskoka Bay
Haliburton
Minden
Bancroft
Orillia
Lake Simcoe
Lake Couchiching
Georgian
Bay
Lake Joseph
Lake Rosseau
Lake Muskoka
Lake Vernon
Peninsula Lake
Fairy Lake
Skeleton Lake
Leonard Lake
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Buck Lake
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Bear Lake
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Kawagama Lake
Johnson Lake
North Lake
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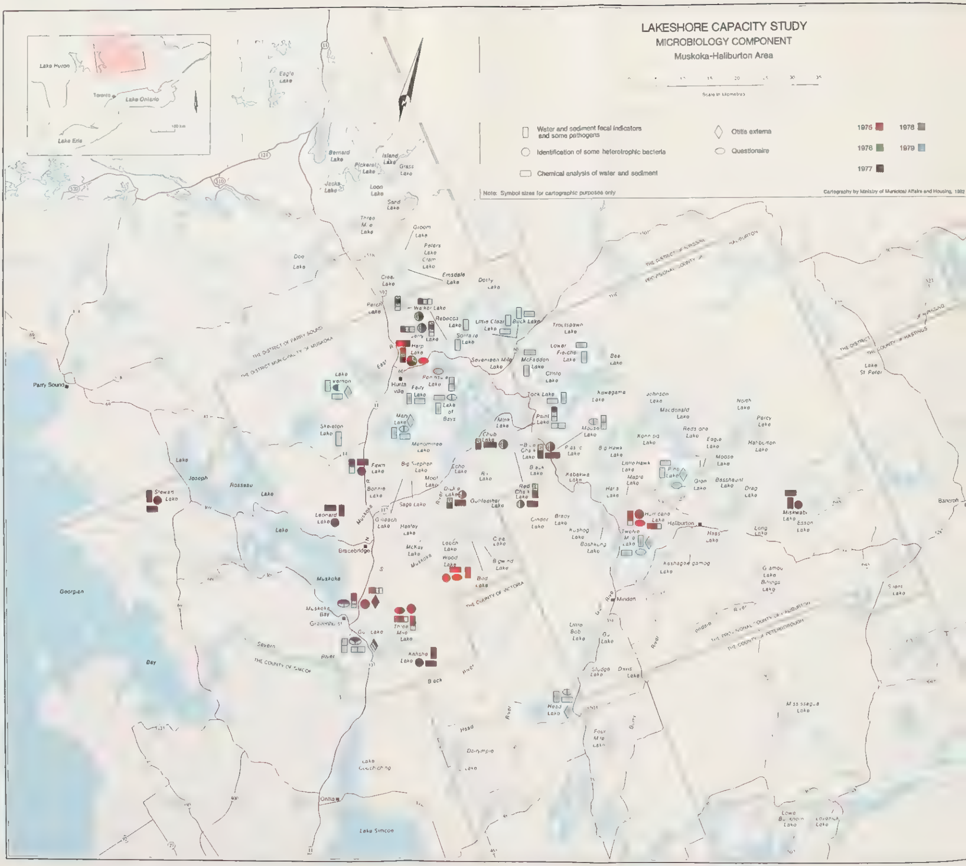
Percy Lake
Kennisis Lake
Eagle Lake
Halburton Lake
Moose Lake
Big Hawk Lake
Little Hawk Lake
Plastic Lake
Mouse Lake
Shoe Lake
Blue Chalk Lake
Black Lake
Red Chalk Lake
Chub Lake
Mink Lake
Paint Lake
Grandview Lake
Martin Lake
Big Stephen Lake
Echo Lake
Fawn Lake
Bonnie Lake
Clearwater Lake
Menominee Lake
Mary Lake
Rill Lake
Wildcat Lake
Gullfeather Lake
Crosson Lake
Pine Lake
Dickie Lake
Kabakwa Lake
Maple Lake
Harvey Lake
Green Lake
Hurricane Lake
Pine Lake
Glen Lake
Basshaunt Lake
Two Islands Lake
Miskwabi Lake
Esson Lake
Long Lake
Head Lake
Brady Lake
Kushog Lake
Boshkung Lake
Twelve Mile Lake
Mountain Lake
Koshlong Lake
Gamour Lake
Billings Lake
Sugar Lake
Dark Lake
Head Lake
Four Mile Lake
Dalrymple Lake
Mississagua Lake
Lower Buckhorn Lake
Lake St Peter
Kahshe Lake
Severn
River
Black River
Gull River
Burnt River
Muskoka River
Gravenhurst
THE DISTRICT MUNICIPALITY OF MUSKOKA
THE PROVISIONAL COUNTY OF HALIBURTON
THE COUNTY OF HASTINGS
THE COUNTY OF PETERBOROUGH
THE COUNTY OF SIMCOE



LAKESHORE CAPACITY STUDY
MICROBIOLOGY COMPONENT
Muskoka-Haliburton Area
Water and sediment fecal indicators and some pathogens
Identification of some heterotrophic bacteria
Chemical analysis of water and sediment
Otitis externa
Questionaire
1976
1978
1976
1979
1977
Note: Symbol sizes for cartographic purposes only
Lake Huron
Toronto
Lake Ontario
Lake Erie
Parry Sound
Georgian
Bay
Huntsville
Bracebridge
Gravenhurst
Muskoka
Haliburton
Minden
Bancroft
Orillia
Lake Simcoe
THE DISTRICT OF PARRY SOUND
THE DISTRICT MUNICIPALITY OF MUSKOKA
THE COUNTY OF VICTORIA
THE COUNTY OF SIMCOE

FOREWORD

The Lakeshore Capacity Study was undertaken to provide a planning tool to assist in managing the development of Ontario's inland lakes. Basic to this task was the need to develop an improved understanding of the relationships between cottage development on the lakeshore and selected aspects of the environment. To accomplish these objectives the Ministry of Municipal Affairs and Housing, responsible for the Study, worked with the ministries of the Environment and Natural Resources.

The Muskoka-Haliburton area of central Ontario was chosen as the study area. This area lies within one physiographic region, part of the Precambrian shield, with similar soils and plant communities. The homogeneity reduced the need to account for major natural variations among the lakes and watersheds. Further, the extent of existing development on the lakes varied; permitting an examination of situations extending from no development to ''full'' development.

The Study involved measurement of the source of the environmental impact, the cottages and their use, and how development affects the indicators of impact: nutrient enrichment; public health; fish, angling and littoral zone; and wildlife and habitat modification. The research findings were linked in a simulation model. The model can predict trends for the various impacts on the watershed.

The Microbiology report examines the relationship of beach use to risk of ear infection. The bacterium *Pseudomonas aeruginosa* causes the ear infection and is transmitted by the swimmers through water, from ear to ear. The major achievement has been the development of quantitative cause-and-effect relationships between swimming and *P. aeruginosa* levels.

The objective of this phase of the Lakeshore Capacity Study, to develop a practical planning tool for lake-watershed management, has been achieved. The next step is further testing prior to implementation.

M.H. Sinclair
Chairman
Lakeshore Capacity Study

ACKNOWLEDGEMENTS

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Ministry of the Environment

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L. Johnston	T.Kafal

Muskoka-Haliburton area medical clinic staff and local physicians

C.A. Burger,
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Microbiology Component

PEER REVIEW

This report was reviewed for the Steering Committee by a panel of specialists:

Dr. V. J. Cabelli	Microbiology	University of Rhode Island North Kingston, Rhode Island	1977 1981
Dr. A. Dufour	Microbiology	U.S. Environment Protection Agency Wakefield, Rhode Island	1981
Dr. P. Seyfried	Microbiology	University of Toronto Toronto, Ontario	1978 1981
Mr. B. Dutka	Microbiology	Canada Centre for Inland Waters Burlington, Ontario	1981
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SUMMARY

The purpose of the Microbiology Component of the Lakeshore Capacity Study was to examine and quantify the relationships among lakeshore development and the recreational activities, the microbiological quality of the lake waters and the incidence of disease among the human users of the water. Between 1976 and 1979, studies were carried out in the Muskoka-Haliburton area of Ontario to determine the water quality of a number of lakes and a few public access beaches and to assess the incidence of human illness. The work centered around the bacterium, *Pseudomonas aeruginosa* and its relationship to an external ear infection called *otitis externa. P. aeruginosa* was found to be absent from or in very low levels in the surface waters. When it was found, it was usually found in association with human activities, such as swimming and the levels found were highly variable over both time and space. The levels of *P. aeruginosa* present in surface waters were related to the water temperature and to the amount of human use, e.g. *P. areuginosa* levels increased as the number of swimmers in a swimming area increased. The availability of nutrients for growth may have affected the levels of *P. aeruginosa* present in surface waters.

During a summer, eight to twelve swimmers out of a population of 100 contracted an ear infection. The incidence of ear infection was seasonally variable with most infections occurring in July and August. Swimmers in the 5 to 24 year age group, those that swam very frequently and those with a previous history of ear infections, were the most likely to contract an ear infection. The incidence of ear infections was related to the level of *Pseudomonas* present in the water of the swimming area. Noticeable effects on the incidence of ear infection were seen as the levels of *P. aeruginosa* in the water showed a median level of 0 per 100 mL, i.e. no *Pseudomonas* was detected, and a 75 percentile level in the range 10 to 100 per 100 mL.

The model developed and the statements made as a result of this study suggest a water quality criterion and design criteria which should be used to reduce the risk of ear infection. Design criteria for water and sewage processes in a development were also recognized.

1. INTRODUCTION AND BACKGROUND

Throughout the four years of field operations the Microbiology Component undertook the task to state quantitatively the public health significance of waterborne bacteria whose populations are most apparently influenced by human usage of the recreational lake environment. The use of water for recreational or drinking purposes was to be related to the risk of disease for the human user associated with lakeshore developments.

1.1 BACTERIAL WATER QUALITY

Because of the numerous uses of water by humans, it is not surprising that water has been directly and indirectly involved with numerous outbreaks and single cases of human disease (Craun 1978a,1979b). Disease occurs when pathogenic or disease-causing microorganisms are ingested, gain entry into a cut or wound, or invade any of the mucous membranes of the human body and water provides a means of transport for the microorganism. While the importance of other microorganisms (fungi, algae, protozoans and viruses) to public health, is acknowledged, this study concentrated upon examining the bacteriological quality of water of recreational lakes.

The quality of surface waters used for drinking and recreational purposes is degraded by the introduction of pathogenic bacteria which are released from the body via the feces. Groups of bacteria, called indicators (total coliforms, fecal coliforms and fecal streptococci) occur at higher concentrations in the intestinal tract, feces and sewage than the pathogenic bacteria. In water above a specified concentration, indictators suggest, firstly, that recent contamination with fecal materials may have occurred, and secondly, that the presence of disease-causing microorganisms is likely to produce an unacceptable risk of disease. Therefore, the greater the concentration of fecal indicators the greater is the likelihood of the presence of pathogenic microorganisms and greater is the risk to health of the users of the contaminated waters. The current guidelines for the assessment of bacteriological water quality (Ministry of the Environment 1978a) have been formulated using this relationship.

However, the aquatic environment is usually hostile to the indicator and pathogenic bacteria. The stresses which fecal bacteria encounter in water (Daubner 1975) will eventually decrease their numbers. In practical terms, therefore, the levels of indictor bacteria and pathogens detected in water will usually decrease with time and distance from the pollution source. This means that any public health significance is both site and time specific. For example, a pollution discharge near one end of a long beach may cause a problem at that end of the beach but not at the other end. If the discharge is stopped, the problem would probably disappear over time.

The expected relationship between indicator densities and pathogen densities also change with time and distance. Differential rates of survival in water allow some of the microorganisms found in fecal material to survive for longer periods than others (Geldreich and Kenner 1969) particularly when the water contains relatively high levels of organic nutrients.

Prolonged survival of indicator and of pathogenic microorganisms has frequently been reported in enriched waters and in aquatic sediments (McFeters and Stuart 1972; Goyal et al 1977; Smith et al 1978; LaBelle et al 1980), and in some waters fecal bacteria have been shown to be capable of regrowth (Gorden and Fliermans 1978). Thus, enriched waters and aquatic sediments are potential reservoirs of feces-derived microorganisms which might be resuspended by environmental factors such as wave action and by recreational activities such as swimming and motor-boating (Geldreich 1970; Gerba and MacLeod 1976; Goyal et al 1977). In areas subject to sporadic sewage discharge, this potential reservoir will appear as the presence of pathogenic mircoorganisms, not predicted by the densities of indicator bacteria (Cabelli 1978). In practical terms, sewage pollution could periodically exert a localized detrimental influence on the water quality of bathing beaches and shoreline areas long after the water quality effects of the original pollution have disappeared from surface water. For these reasons surface waters should never be used for drinking purposes without the benefit of adequate disinfection (Ministry of Environment 1978b).

The microbial populations of the lakes and other surface waters are comprised of numerous genera and species. With the exception of a few recently reported species (*Aeromonas hydrophila*, *P. aeruginosa*, *Yersinia enterocolitica)* most bacteria in water are considered to be harmless, normal inhabitants of the aquatic environment. In the natural setting these bacteria are of little or no public health significance although they may be of vital importance in the food webs (nutrient cycling, decomposition) and self-purification of lakes (Wetzel 1975). Nutrient-enrichment or thermal alteration of the environment as a result of human activity or natural calamity, creates an imbalance in the bacterial populations. Increased bacterial densities may result in conditions such as oxygen depletion, fish disease epidemics, and growth of potentially pathogenic bacteria such as *Aeromonas hydrophila* (Cabelli 1973, 1978; Cook et al 1978). While considerable evidence for the involvement of waterborne bacteria in human disease exists (Craun 1978a, b; Pipes 1978), there is currently a lack of conclusive evidence concerning the significance to human health of pathogens which grow in the aquatic environment in response to pollution. Some evidence exists for the involvement of human pathogens in the disease of fish exposed to polluted waters (Hazen 1979; Robohm et al 1979). The

growth of some of these bacteria in response to nutrient or thermal pollution may also be of human health significance. In recreational lakes where pollution sources are usually quite diffuse, the bacterial response to nutrients might be very difficult to determine until the entire lake, rather than specific sites, is affected. Near major sources of pollution, however, these effects might be readily detected particularly in seasons of warm water temperatures and in sheltered bays.

1.2 PUBLIC HEALTH SIGNIFICANCE AND USE

Previous studies have provided little quantitative information about relationships between recreational activity and bacterial densities in water and virtually no information about the incidence of human illnesses. Several studies have concluded that gross recreational activities had little or no influence upon bacteriological water quality of lakes or watersheds (Rosebery 1964; Walter and Bottman 1967; Carswell et al 1969). These studies did indicate, however, that localized water quality degradation occurred on shoreline receiving intensive recreational activity during the summer season. The increased bacterial levels observed at sites such as bathing beaches had no long-term influence on water quality of either the most affected sites or on the lake or watershed as a whole. Other reports, however, concluded that recreational use of reservoirs or watersheds intended for supply of municipal waters caused deterioration of the water quality of both localized site and of the whole body of water (Karalekas and Lynch 1965; Kleemola 1971; Stuart et al 1971; Dietrich and Mulamottil 1974; Skinner et al 1974; Wagenet and Lawrence 1974; Baker and Magnuson 1976). A recent study suggested both a degradation of water quality and the existence of a potential health hazard in downstream reaches of a watershed during periods of intensive human use of upstream camping areas (Varness et al 1978).

These conflicting conclusions concerning the influence of recreational activities may be attributable to the failure of most studies to take into account the effect of countermeasures (e.g. septic system, disinfection), to quantitatively determine the effect of the populations of animals (Walter and Bottman 1967; Lee et al 1970), the degree of dispersion of recreational activities (Varness et al 1978), and the degree of human use (Lee et al 1970) in each watershed examined. For instance, although Lee et al (1970) detected pathogenic bacteria in the downstream reaches of each watershed, they were unable to show a quantitative influence of human usage upon bacterial indicator populations in any of the watersheds. The study employed a measure of human use (man days per square mile) and wildlife (game animals per square mile) in an attempt to determine the influence of human use levels on the bacteriological quantity of 3 watersheds, only one of which was protected from recreational use. They concluded that since the game animals use was estimated to be much higher than the human use in each watershed, the influence of the humans was not detectable with current methods due to the dominant influence of the animals. In these watersheds animals were considered to be the major contributors of pollution which resulted in the fecal coliform densities observed. They recommended that watersheds with relatively higher levels of human use be examined to determine what level of human use would influence indicator bacterial densities. Unfortunately this recommendation has not been incorporated in subsequent studies of relationships between land use and water quality. With the exception of a few site specific investigations (Cabelli et al 1976, 1979; Fraser 1977), little information exists concerning the incidence of disease among human users of recreational waters. Nevertheless three major aspects of lakeshore development and land use can be considered to be very important to the manifestation of disease among human users of recreational lakes. These are: 1) use of water for drinking; 2) use of sewage systems; and 3) use of water for recreation, particularly body-contact activities such as swimming.

1.3 DRINKING WATER AND ILLNESS

Users of lakeshore developments require water of good quality for drinking, food preparation, personal hygiene, and for other domestic purposes. In the past decade numerous outbreaks of human illnesses, particularly gastroenteritis, have been attributable to the drinking of untreated ground and surface waters amd waters from supplies with deficient treatment systems (Craun 1978a, b; 1979b; Pipes 1978). Infections caused by bacteria (cholera dysentery, typhoid, tuberculosis), viruses (hepatitis, polio) and parasites (trinchinosis, amoebiasis, lambliasis) are frequently waterborne. But the majority of outbreaks and cases often involved drinking waters for which no causal organism could be conclusively demonstrated (McCabe 1977). Many of these latter outbreaks involving gastroenteritis, however, were probably caused by three recently recognized pathogens, *Yersinia enteroclitica*, enteropathogenic *Escherichia coli* and *Giardia lamblia* (Craun 1979a). Each of these are frequently encountered in surface waters. Because of the possible presence of pathogenic microorganisms in surface waters and because of the frequent fecal contamination of ground waters by leachates from septic tank systems (Sandhu et al 1979), water supplies intended for drinking purposes must be treated in some manner in order to protect the health of the consumers. This is especially significant to cottagers and recreationists whose water supplies are most often lakewater or whose wells are situated on lakeshore lots close to septic tanks or to potentially contaminated surface waters. The risk of waterborne illness can be markedly reduced by simple but important sanitary considerations: the boiling or other thorough disinfection of surface waters before consumption; and the proper installation and maintenence of wells and sewage systems (Ministry of Environment 1978b).

1.4 SEWAGE TREATMENT AND ILLNESS

Production of fecal material is an inherent consequence of the human or animal presence in the recreational lake environment. Sewage pollution resulting from improper storage, treatment or disposal of fecal waste materials, or from excessive discharges of treated sewage, can have serious detrimental influences upon the ecology and water quality of lakes and rivers (Hynes 1960; Piecynska 1975; Bell et al 1976; Jumppanen 1976; Gibbs 1977; Ministry of Environment 1978b) and on ground water quality (Craun 1979b; Sandhu et al 1979; Goyal et al 1980).

Considerable advances have been made in controlling the undesirable influences of sewage pollution related to the eutrophication of lakes (Dillon and Rigler 1975; Jorgensen 1976; Burton and King 1979) but from the public health stand-

point sewage pollution continues to be involved in many outbreaks and cases of waterborne human illnesses (Craun 1979b). Sewage pollution has been associated with bacterial and viral infections contracted by bathers using recreational waters (Denis et al 1974; Rosenberg et al 1976) but is more frequently associated as the cause of outbreaks involving contaminated drinking waters. For instance, from 1971 to 1977 there were 192 outbreaks and 36,757 cases of illness related to the consumption of contaminated drinking waters in the United States (Craun 1979c). Overflow from septic tanks and cesspools accounted for 42% of those waterborne outbreaks. Septic tank systems, as employed by most cottagers usually must handle a combined sewage and domestic wastewater load of about 126 litres per user per day (Brandes 1978). Because of the high nutrient and bacterial content of this wastewater, the septic systems must be properly designed, installed and operated to prevent contamination of surrounding ground and surface waters. Although septic systems may contribute to water quality degradation (Gibbs 1977), their impact on bacteriological quality of lake waters seems to be limited to areas nearest defective or substandard systems (Hendry and Toth 1980). Major impacts on water quality associated with sewage pollution therefore may be qualitatively and quantitatively observed at point sources, particularly those near outfalls of larger treatment systems which serve developments with higher human use levels. Regardless of source, however, sewage pollution is the major contributor of infectious disease microorganisms to drinking and recreational waters (Cabelli 1978; Craun 1979b).

1.5 RECREATIONAL USE AND ILLNESS

Reports of outbreaks of human illness associated with swimming have been relatively uncommon although several serious outbreaks of bacterial and viral infections due to swimming have been reported (Rosenberg et al 1976; Davis et al 1974; Pipes 1978). A correlation between the incidence of gastronintestinal illness among swimmers and fecal coliform (*Escherichia coli*) densities has been demonstrated at marine bathing beaches (Cabelli et al 1976, 1979). The human diseases associated with swimming have also been from wound infections by *Aeromonas hydrophila* (Rosenthal et al 1974; Hanson et al 1977) and infections of the eyes, ears, nose, skin and throat by a variety of bacteria and viruses. Of these, the ear infection, *otitis externa* or ''swimmer's ear,'' caused by the bacterium *P. aeruginosa* (Jones 1965; Cassisi et al 1977), accounts for the majority of reported swimming-related illnesses. The bacterial indicator systems usually employed to determine water quality and health significance of drinking waters, often are unreliable in predicting the presence of pathogentic microbes in recreational waters and the health risks associated with swimming (Foster et al 1971; Dudley et al 1976; Geldreich 1976; Cabelli 1978; Pipes 1978; Reasoner 1978; Cabelli et al 1975, 1976, 1979). Several reports (Cabelli 1978; Pipes 1978) have recommended the use of *P. aeruginosa, A. hydrophila* and other quantifiable microbes of health significance as new indicator systems for assessing recreational water quality. *P. aeruginosa* has been included as an indicator of a potential health hazard in bathing waters in Ontario (Ministry of Environment 1978a).

The infectious microoganisms found in recreational waters can come from one of three sources, pollution by human or animal sewage, the washing of the skin of the bathers (Robinton and Mood 1966; Hanes and Fassa 1971) or the resident population of environmental microbes. The latter of these sources is generally not controllable but seems to play a minor role in the causation of human disease. For either of the other two sources, the contribution of the infectious agent to the recreational waters is dependent on factors such as the level of the illness in the contributing population, the number of carriers of the agent and the effective countermeasures taken (Cabelli 1978). As a result, if large human populations are present, the probablity of encountering ill or diseased individuals or carriers is very high and therefore, the occurrence of the infectious organism in the source is almost a certainty. However, with very small populations, such as the bathers at a single beach or the residents of a group of cottages, the probability of an ill individual or a carrier being present in the population is correspondingly very small. Under these circumstances, the occurrence of the infectious organism will be sporadic and the usual indicator systems may fail. This failure arises from a breakdown in the indicator-pathogen relationship on which the indicator system depends. Determination of the concentration of pathogen directly alleviates this problem but is generally more costly and is technically more difficult, if not impossible, depending on the infectious agent.

Sources of infectious agents which originate from sewage can be effectively removed by appropriate countermeasures such as adequate treatment and disinfection. However, the direct contribution of infectious agent by the bathers themselves through the washing of skin has largely been ignored and is poorly understood. Effective countermeasures for this latter source are generally not possible.

1.6 *PSEUDOMONAS AERUGINOSA — OTITIS EXTERNA*

This study was set up to examine the recreational water quality as it relates to the bacterium, *P. aeruginosa* and the ear infection, *otitis externa* . *P. aeruginosa* is a bacterium known to cause skin rashes (Kush and Hoadley 1980) and eye infections (Wilson and Ahearn 1977) and is the primary organism associated with the ear infection, *otitis externa* (Alcock 1977; Cassisi et al 1977). It appears to be ''associated primarily with man'' and ''slightly in excess of 10% among healthy adults'' in the United States are intestinal carries of *P. aeruginosa* (Hoadley 1977). It can therefore be routinely in human sewage and has been proposed as an indicator of sewage pollution (Cabelli 1977). As an infectious agent *P. aeruginosa* is described as an opportunistic pathogen. The organism may be present without apparent illness or disease but if the resistance of the host has been reduced by trauma, other diseases, medication or predisposition, a secondary infection by *Pseudomonas* can occur.

Although a direct correlation has not been established for ear infections due to waterborne *P. aeruginosa* , the reported incidence of *otitis externa* is higher among swimmers than non-swimmers (Hoadley and Knight 1975). Ear infections due to this bacterium may occur among swimmers even when low densities of *P. aeruginosa* are present in the bathing water (Seyfried and Fraser 1978).

1.7 *PSEUDOMONAS — OTITIS* MODEL

A model for the relationship of such factors as the chemical and physical environment, and human use and activity with *P. aeruginosa* and the illness, *otitis externa* is presented as a flow diagram in Figure 1. This model is supported by the general microbiology literature and by the general inclination of microbial ecology.

P. aeruginosa along with many other bacteria is a normal inhabitant of the human gastro-intestinal system (Hoadley 1977). These bacteria aid in the digestion of food by the human and are excreted in large numbers with the feces or waste products of human digestion (Boxes A and B). The bacteria including the *Pseudomonas* will, therefore be found in sewage and the normal treatment processes associated with sewage. Through this mechanism, *Pseudomonas* can be introduced to the environment, in particular, to the water. In addition, the washing of the human body which occurs during swimming activity will introduce *Pseudomonas* and other bacteria into the water (Hanes and Fossa 1971). These bacteria will come from the entire skin surface of the human but in particular the anal area.

Once in the water, the *Pseudomonas* and other bacteria whether they came from sewage or skin washings can be reintroduced into the human population through the use of the water for drinking, washing or food preparation purposes (Box D).

The survival and growth of *Pseudomonas* in the environment is dependent upon a number of factors including the water temperature, the chemical nutrients available and the interactions which occur between microorganisms (Box E). Hoadley (1977) suggests that in ''unpolluted but organically enriched surface waters when water temperatures exceeded 30°C, growth might occur when the temperatures were high.'' At temperatures lower than 30°C, *Pseudomonas* is not growing but longer survival will occur. The definition of ''organically enriched surface waters'' has yet to be explored and what chemicals constitute organic enrichment for a particular bacterium are not known in detail. Hoadley (1977) does however add that *P. aeruginosa* ''does not normally inhabit northern temperature surface waters unless recently affected by human activity or the activities of domestic animals.''

The contribution of animals to the cycling of *P. aeruginosa* appears to be largely as a passive carrier rather than a reservoir (Hoadley and McCoy 1968). Therefore, their contribution will be minimal (Box F).

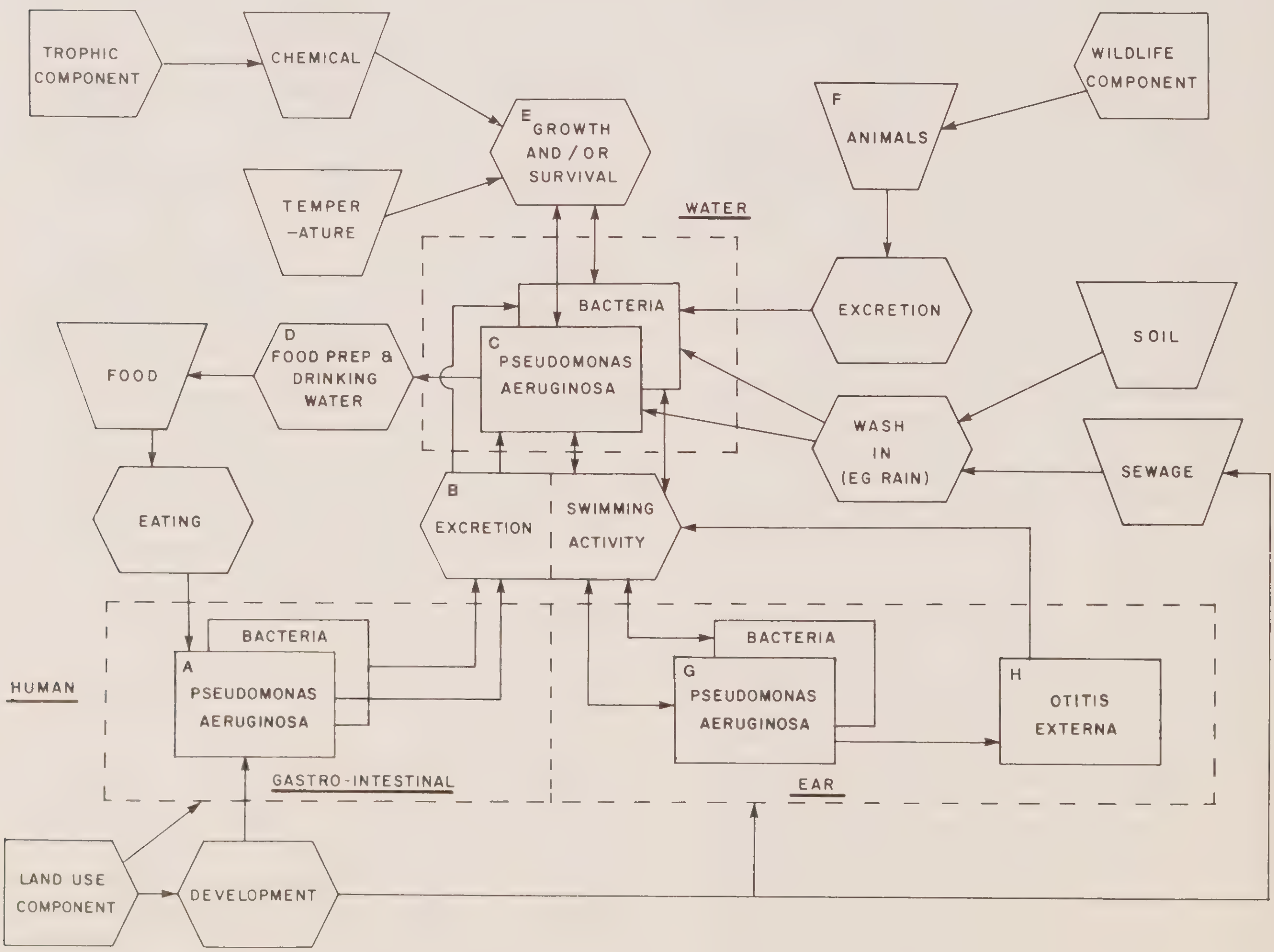


Figure 1: Flow diagram for *P. aeruginosa* and its relationship to swimming activity and the ear infection, *otitis externa*.

The portion of the model which this study examined most intensively was the linkages between *Pseudomonas* in the water, swimming activity, and *Pseudomonas* in the ear of a human user.

When a person goes swimming in the water which contains *P. aeruginosa* and the person submerges their head under the water, a small but definable portion of water enters the ear canal carrying the *Pseudomonas* with it (Box G.) This water with its bacterial content may drain out or be dried out without any further action or may moisten the ear canal contents and remain for some considerable time. The bacteria may also settle out or adhere to the inner surfaces, even though the water may be removed, leaving behind the bacteria. Once the *Pseudomonas* has gained access to the ear, an ear infection can occur (Box H). The process by which the infection occurs is poorly understood. However the people in the age group 5 to 15 years, the people who swim very frequently during any month and the people who have experienced a previous ear infection seem to be more likely to contract an ear infection. In either the case where *Pseudomonas* is in the ear with no apparent illness or the case where *Pseudomonas* has caused an ear infection, further swimming activity by that individual may reintroduce the *Pseudomonas* back into the water environment at the location of the swimming activity. Hoadley and Knight (1975) examined the ears of both healthy individuals and those with an ear infection. *P. aeruginosa* was found in 78% of the swabs from infected ears of swimmers and 33% of the swabs from infected ears of non-swimmers. The corresponding figures for a control population without ear infections were 11% and 14% respectively. They found a seasonal effect on the frequency of isolation of *Pseudomonas* from ears. Their telephone survey showed that 7.3% of the general swimmer population and 3.1% of the non-swimmers experienced earaches during the summer season. Seyfried and Fraser (1978) showed a higher isolation of *P. aeruginosa* and a higher incidence of otitis externa ear infection among the subjects using a swimming pool with a higher contamination with *Pseudomonas.* In both studies, the levels of *P. aeruginosa* in the water were low, 0 to 100 per 100 mL, and were very highly variable.

2. STUDY AREA

During the course of the Lakeshore Capacity Study, twenty-four lakes in the Muskoka-Haliburton area of Ontario were examined for bacterial and chemical water quality (Table 1). These lakes were examined in sufficient detail to permit comments to be made concerning possible relations between development effects and bacterial water quality. For this report, water quality will be discussed in generalities using specific examples and no lake will be discussed in detail.

Table 1. List of lakes in the study 1976-1979 and the total number of surface samples examined for the bacteria *P. aeruginosa* excluding beach samples.

Lake	Year	NUMBER OF SAMPLES BY SEASON Spring (May-June)	Summer (July-Aug.)	Fall (Sept.-Nov.)
1. Pine	1979	28	56	—
2. St. Mary	1979	42	14	—
3. Seventeen Mile	1979	20	32	—
4. Buck	1979	18	27	—
5. Head	1979	42	84	—
6. Vernon	1979	42	125	—
7. Mary	1979	42	121	—
8. Twelve Mile	1979	14	42	—
9. Walker	1976	—	24	18
	1977	6	56	—
	1978	41	42	—
	1979	42	28	—
10. Muskoka Bay	1977	42	56	41
	1978	83	42	14
	1979	54	42	—
11. Gull	1978	122	126	126
	1979	55	98	—
12. Jerry	1976	—	18	18
	1977	6	56	—
	1978	41	42	28
	1979	2	14	—
13. Three Mile				
a) east half	1977	42	36	—
	1978	40	—	—
b) west half	1977	42	56	58
	1978	56	27	—
14. Hurricane	1977	5	42	—
	1978	28	56	—
15. Harp	1976	—	18	18
	1977	6	52	—
16. Fawn	1977	6	53	—
17. Chub	1976	—	18	17
	1977	6	—	—
18. Dickie	1976	—	18	18
	1977	6	—	—
19. Kahshe				
a) east half	1977	6	—	—
b) west half	1977	6	—	—
20. Stewart	1977	6	—	—
21. Red Chalk	1976	—	18	18
	1977	6	—	—
22. Blue Chalk	1976	—	18	18
	1977	6	—	—
23. Leonard	1977	6	—	—
24. Miskwabi	1977	6	—	—

Initially, the lakes studied by the Microbiology Component were chosen from among those also studied by the Trophic Status Component and had varying numbers of cottages or other forms of development along the shoreline. A few lakes (e.g. Jerry Lake) were undeveloped with respect to cottages and human usage. The cottages were organized as mainly single-tier cottage developments. The degree of dispersal of cottages along the shorelines varied. The undeveloped shorelines were mostly forested but some were used for agricultural rather than recreational purposes. Road access to the lakes and cottages was usually available. The cottages were serviced by septic tank sewage treatment systems. Lake water, with a varying amount of treatment, was the usual form of drinking water supply. A few lakes (e.g. Muskoka Bay and Gull Lake) were situated in urban areas with municipal sewage and water treatment plants. There were one or more marinas and commercial developments such as lodges or resorts.

In 1979, the lakes examined had at least one public access beach which was intensively used for recreation particularly during the summer months. The water quality of specific beaches (Table 2) was also examined. In general, these beaches were about 100 metres wide with a gently sloping sand bottom. The boundaries of the beach swimming areas were usually delineated by floating marker ropes and the area patrolled by lifeguards. Floating or permanent docks may have been in the vicinity, some serving as diving platforms. These beaches provided sites for organized swimming classes during the week but were also available to the general public at all times.

Table 2. List of beaches studied as part of the Lakeshore Capacity Microbiology Component.

Beach	Lake	Nearest Town	Year
No name	Gull	Gravenhurst	1977, 78, 79
Rotary	Head	Haliburton	1979
Lion's	Mary	Port Sydney	1979
No name	Pine	West Guilford	1979
Kinsmen	Vernon	Huntsville	1979
Public	Twelve Mile	Carnarvon	1979
Red Cross	Twelve Mile	Carnarvon	1979

At each beach there was an associated dry-beach sand area and a treed park and picnic area. The study beaches represent a large number of the public access beaches in the Muskoka-Haliburton area. This report will be examining beaches as a class of recreational use of the small lakes in the Muskoka-Haliburton area.

Figures 2, 3 and 4 are sketch maps of Muskoka Bay, Gull Lake and Head Lake, respectively, showing the locations at which water or sediment samples and some general

geographical information were obtained. Figures 5 and 6 are sketch maps of Gull Lake beach and Head Lake beach showing the sampling locations and the general relationship of the swimming areas. Each beach area was divided into quarters or sectors; each having 3 or 4 sampling locations associated with it. These figures have been provided as examples of study maps and do not provide exhaustive information or show all locations examined.

Figure 2: Sketch map of Muskoka Bay showing sampling locations and major geographic features.

Figure 3: Sketch map of Gull Lake showing sampling locations and major geographic features.

Figure 4: Sketch map of Head Lake showing sampling locations and major geographic features.

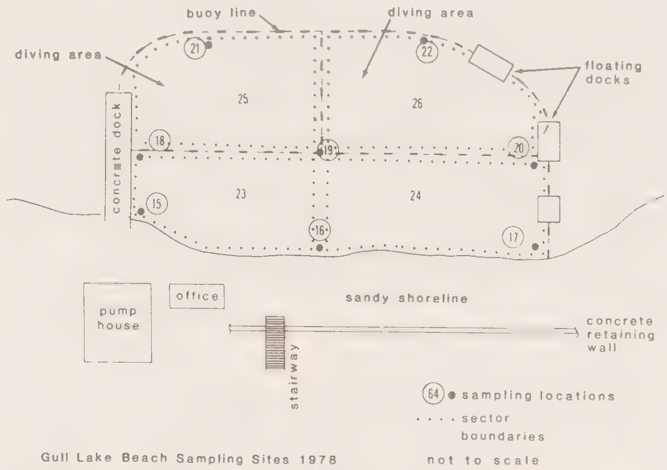


Figure 5: Sketch map of Gull Lake beach in Gravenhurst showing sampling locations and major structural features. Beach sampling locations were numbererd from 15 to 22 consecutively. Sectors were assigned the numbers 23 through 26.

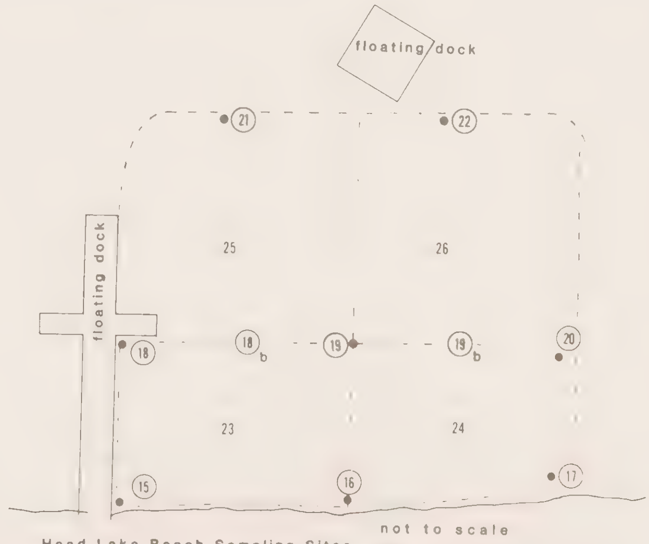


Figure 6: Sketch map of Head Lake beach near Haliburton showing sampling locations and major structural features. Beach sampling locations were numbered from 15 to 22 consecutively. Sectors were assigned the numbers 23 through 26.

3. METHODS

The field work conducted by the Microbiology Component involved two types of surveys. One consisted of regular sampling surveys of lakes to determine the distribution and densities of bacteria and levels of important nutrients. These regular sampling surveys were designed to provide information concerning the effects of development on bacteriological water quality and upon possible relationships between development, bacterial densities and nutrient status. The other type of survey entailed an epidemiological investigation of *P. aeruginosa*-related outer ear infection (*otitis externa*) among users of the shoreline waters and beaches of recreational lakes.

3.1 BACTERIAL AND CHEMICAL SAMPLING

During regular sampling surveys, which were conducted during the spring (May to June), summer (July to August) and fall (September to November) seasons, shoreline water samples were collected at a depth of one metre below the surface at established sampling locations on each study lake (Table 1). These samples were analyzed by membrane filtration procedures for indicator bacteria and for *P. aeruginosa* (Table 3).

Table 3. Microbiological analysis methods used in the Lakeshore Capacity Study (Ministry of Environment 1980).

Analysis	Medium	Temperature (°C)	Incubation time (hr)
a) *Membrane filtration*			
Total coliform (TC)	m-Endo LES agar	35	22
Fecal coliform (FC)	MacConkey membrane broth	44.5	20
Fecal streptococcus (FS)	m-Enterococcus agar	35	48
P. aeruginosa (Psa)	mPA agar	41.5	48
b) *Spread plate*			
Heterotrophic bacteria (HB)	Foot and Taylor agar	20	168
c) *Most Probable Number (MPN)*			
Fecal coliform	Lactose broth	35	48
(FC)	EC broth	44.5	24
P. aeruginosa (Psa)	Drake's broth	41.5	96
	Pseudocel agar	35	24
	King's A agar	35	24

Sediment samples were collected concurrently from the lake bottom at the same shoreline locations. These samples were analyzed by the most probable number tube-dilution methods for indicator bacteria and for *P. aeruginosa*. Composite water samples obtained from mid-lake sampling locations, were analyzed for indicator bacteria, *P. aeruginosa*, heterotrophic bacteria and for various chemicals. Generally, these samples were examined for: total Kjeldahl nitrogen, dissolved Kjeldahl nitrogen, nitrate, nitrite, ammonium, total phosphorus, total carbon, total organic carbon, inorganic carbon, pH, conductivity, colour and chlorophyll. At each sampling the secchi disk, temperature and dissolved oxygen concentration of the water at each location was recorded.

3.2 BEACH STUDIES

In 1977 a pilot project to determine the incidence of *P. aeruginosa*-related ear infections among members of organized swimming classes at Gull Lake beach and among cottagers along the shore of Muskoka Bay was initiated. In the two successive years this project was greatly intensified and additional public beaches at lakes near major communities in the study area were included as study sites. Information was gathered related to the histories of ear infection, swimming habits and the water-based recreational activities of cottagers, swimmers and non-swimmers. Physicians and hospital emergency ward staff in communities nearest the study beaches were encouraged to cooperate by providing medical information about each case of *otitis externa* treated and by supplying bacterial swab samples from the ears of their *otitis externa* patients. Participating physicians distributed questionnaires to these patients to obtain information about each patient's swimming habits and experiences with ear infection. Through personal interviews, mail-in and telephone questionnaires, information was gathered at the start and end of the swimming season from cottagers, swimmers and non-swimmers at public beaches and from medical personnel. Examples of questionnaires and relevant explanatory letters used in this study are attached as Appendix A.

Extensive sampling of the bathing beach waters as well as sampling of shoreline sediments was conducted. These samples were analyzed for indicator bacteria and *P. aeruginosa* as described for lake surveys. On each sampling day at the beaches during 1977 and 1978 the ears of swimming class members were swabbed both prior to and just after their swimming lesson. These swab cultures were analyzed for the presence of *P. aeruginosa*. Sampling of the beach waters was carried out at established stations within the designated swimming areas at intervals on each sampling day before and after known numbers of swimmers had entered the water. Air and water temperature at each sampling location and counts of the number of swimmers in the water and on the dry beach at hourly intervals were also obtained.

Information obtained from lake and beach surveys were analyzed to determine relationships between incidence of *otitis externa*, bacterial and chemical water quality, levels of human use (bather load) of water, and environmental conditions.

3.3 FURTHER IDENTIFICATION OF *P. AERUGINOSA*

Isolates of *P. aeruginosa* were retained from selected samples from water, sediment and swimmers' ears and characterized by serological typing procedures (Ministry of Environment 1980). The tube procedure employed for the testing of sediment samples and ear swabs did not give rise to distinct colonies. Therefore only one isolate was retained for each tube yielding a positive result for *P. aeruginosa*.

At present, serotyping is the most reliable method which differentiates members of a particular bacterial species on the basis of minute structural differences (Brokopp and Farmer 1979; Edmonds 1972). The serotyping procedure employed in this study led to the subgrouping of the isolates on the basis of their cell wall structures. Each trait on the cell wall has been given an assigned code, usually a number, a few letters, or a combination of both. The isolate, which possessed a particular trait, was placed in a subgroup called a 'serotype' which was identified by the trait code. Often bacteria were found to exhibit two or more traits and then they were given a serotype to indicate each trait detected.

A total of 2222 cultures of *P. aeruginosa* were retained during 1977 and 1978 for serotyping (Table 4). In 1977, of 344 cultures from lake and beach study sites, 141 originated from Gull Lake beach and 161 from Muskoka Bay. In 1978, a total of 1824 cultures were retained. Again, the majority of these originated from Gull Lake beach.

Differences in the sampling programs employed in the two years account largely for the differences seen in the numbers of isolates from each study site and source. The sediment sampling program in 1977 led to the recovery of a greater number of isolates from the sediments in 1977 than in 1978. In 1978, the intensified study at Gull Lake beach led to recoveries of greater numbers of isolates from the beach waters and the ears of swimmers than during the previous year.

In 1977, all nine cultures from samples supplied by participating physicians were acquired from the ears of *otitis externa* patients in the municipality of Gravenhurst (Table 5). In 1978 the scope of the *otitis externa* patient study was expanded into the communities of Haliburton, Huntsville, and Bracebridge. However, this increase in the number of cultures is due partly to the increase in the number of participating physicians in Haliburton, Huntsville, Bracebridge.

In 1977, sufficient materials were purchased from the University of Toronto (Professor P. Seyfried and Mr. D. Fraser) to permit the serotyping of *P. aeruginosa*. Except for a small number of cultures, all isolates retained in 1977 were serotyped. However, only a fraction of the *P. aeruginosa* isolates obtained from lake and beach waters and sediments in 1978 could be characterized serologically.

Of the 2222 isolates retained for serotyping, 1612 were typed, 1149 from water samples, 97 from the ears of swimmers and non-swimmers, and 52 from the ears of *otitis externa* patients who had been treated by area physicians.

Table 4. The number of *P. aeruginosa* cultures isolated and retained from each site studied in 1977 and 1978.

Study Site	Source of Isolation							
	Water		Sediment		Swimmer's Ears		Total	
	1977	1978	1977	1978	1977	1978	1977	1978
Deionized Water	-	1	-	-	-	-	-	1
Dwight Beach	-	5	-	-	-	-	-	5
Gull Lake	-	94	-	14	-	-	-	108
Gull Lake Beach	50	1280	78	81	13	78	141	1439
Head Lake Beach	-	13	-	-	-	-	-	13
Hidden Valley Beach	-	3	-	-	-	-	-	3
Hurricane Lake	0*	7	0	1	-	-	0*	8
Jerry Lake	0	58	0	9	-	-	0	67
Mary Lake	-	10	-	-	-	-	-	10
Mouse Lake	-	11	-	-	-	0	-	11
Muskoka Bay	55	49	101	5	5	-	161	54
St. Mary's Lake	-	11	1	-	-	-	1	11
3 Mile Lake — West	8	6	15	0	-	-	23	6
Vernon Lake Beach	-	29	-	-	-	-	-	29
Walker Lake	0*	38	7	21	-	-	7	59
Fawn Lake	0*	-	2	-	-	-	2	-
Harp Lake	0*	-	4	-	-	-	4	-
3 Mile Lake — East	0*	0*	3	0	-	-	3	0*
Lagoon	2	-	-	-	-	-	2	-
Total	115	1615	211	131	18	78	344	1824

- Study/Source not sampled
0 Study/Source was sampled but *Pseudomonas* was not encountered
0* Study/Source was sampled, *Pseudomonas* was encountered, no cultures retained.

Table 5. The number of *P. aeruginosa* cultures isolated and retained from each physician in 1977 and 1978.

Physician	Patients' Ear Isolates		Total
	1977	1978	
Haliburton Dr. #1	-	9	9
Haliburton Dr. #2	-	2	2
Haliburton Dr. #3	-	8	8
Haliburton Dr. #4	-	3	3
Huntsville Emergency Ward	-	7	7
Gravenhurst Dr. #3	3	1	4
Gravenhurst Dr. #4	2	0	2
Gravenhurst Dr. #6	1	-	1
Bracebridge Emergency Ward	-	12	12
Bracebridge Dr. #2	-	2	2
Bracebridge Dr. #9	-	1	1
Dr. Unknown	3	-	3
Total	9	45	54

- No swabs received

0 Swab received but *Pseudomonas* not encountered.

4. RESULTS AND DISCUSSION

4.1 MICROBIOLOGICAL WATER QUALITY

4.1.1 LAKES AND BACTERIAL INDICATORS

In general, the water quality of the lakes in the Muskoka-Haliburton area of Ontario appears to be good when compared to current microbiological water quality objectives. The water from these lakes should, with minimal treatment, be potable and should also be acceptable for recreational purposes.

In the study lakes, annual median total coliform levels ranged from 0 to 1000 bacteria per 100 mL. These levels compared favourably with the Ontario recreational use objective (Ministry of Environment 1978a) of 1000 total coliforms per 100 mL. However, the total coliform group is questionable as a sanitary indicator (Dutka 1973), and the acceptability of the water quality can not be absolutely proven using this group of bacteria.

The median fecal coliform levels in the study lakes ranged from 0 to 30 bacteria per 100 mL. Again, these levels compare favourably with the Ontario recreational use objectives (100 fecal coliforms per 100 mL). Greater detail about the fecal coliform levels in the various study lakes is presented in Figures 7, 8 and 9. Figure 7 presents the fecal coliform levels at the shoreline locations; Figure 8 at mid-lake locations; and Figure 9 at the inflow and outflow locations.

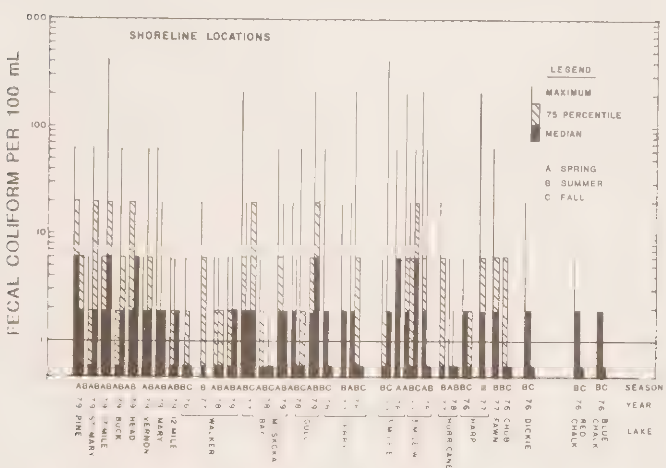


Figure 7: Plot of seasonal median, 75 percentile and maximum levels of fecal coliform per 100 mL for all shoreline locations by lake, season and year. The seasons used here are: spring (A) – May and June; summer (B) – July and August; and Fall (C) – September to November.

A year-to-year variation is suggested in most lakes. For example, Muskoka Bay sampling locations exhibited lower fecal coliform levels in 1978 than either 1977 or 1979. A seasonal variation is also suggested in most lakes with spring (May and June) or summer (September to November) usually having higher bacterial levels than fall (September to November). In general, lake inflows and outflows have the highest fecal coliform levels and the mid-lake locations have the lowest.

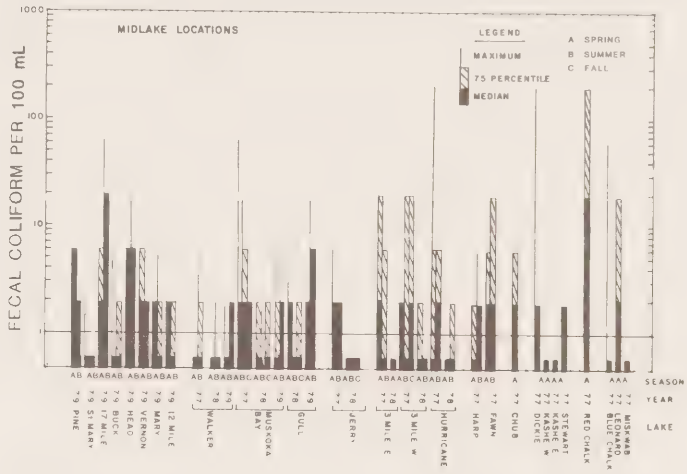


Figure 8: Plot of seasonal median, 75 percentile and maximum levels of fecal coliform per 100 mL for all midlake locations by lake, season and year. The seasons used here are: spring (A) – May and June; summer (B) – July and August; and fall (C) – September to November.

Each of these variations in fecal coliform level and the additional variations between locations, weeks, days and times within a day are expected normal variations for any of the bacteria of public health significance (Bennett 1969).

Total coliform levels (Figure 10) tended to be one to two orders of magnitude higher than the corresponding fecal

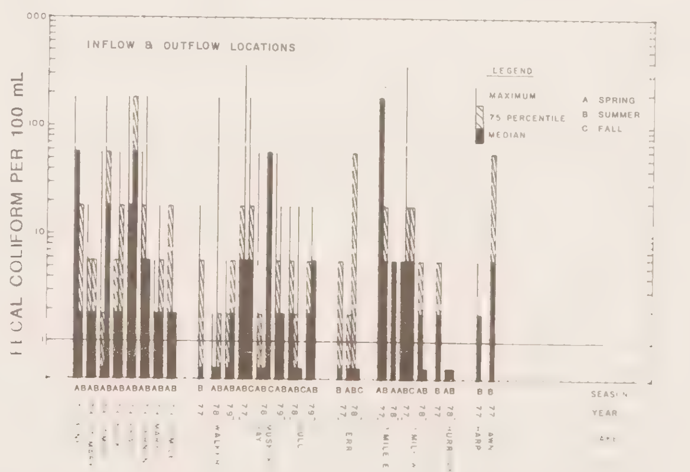


Figure 9: Plot of seasonal median, 75 percentile and maximum levels of fecal coliform per 100 mL for all inflow/outflow locations by lake, season and year. The seasons used here are: spring (A) – May and June; summer (B) – July and August; and fall (C) – September to November.

coliform levels. However, no specific relationship was apparent between total coliforms and fecal coliforms except that, as total coliform levels tended to increase, fecal coliform levels also tended to increase.

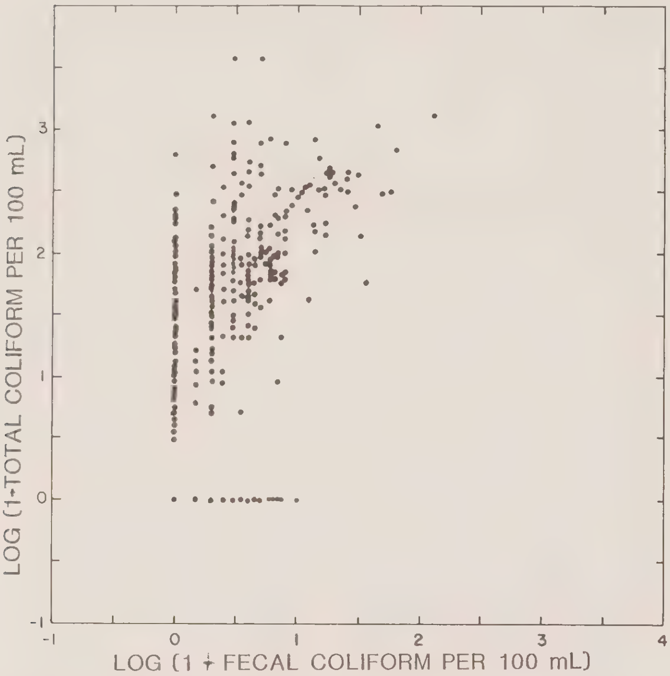


Figure 10: Plot of annual median levels of total coliforms versus fecal coliforms (on logarithmic scales) for all locations on all lakes for the years 1976 to 1979. Each point is from one location on a lake for a year.

The median fecal streptococcus levels in the study lakes ranged from 0 to 100 per 100 mL. Fecal streptococcus levels (Figure 11) tended to be approximately one order of magnitude higher than the corresponding fecal coliform levels.

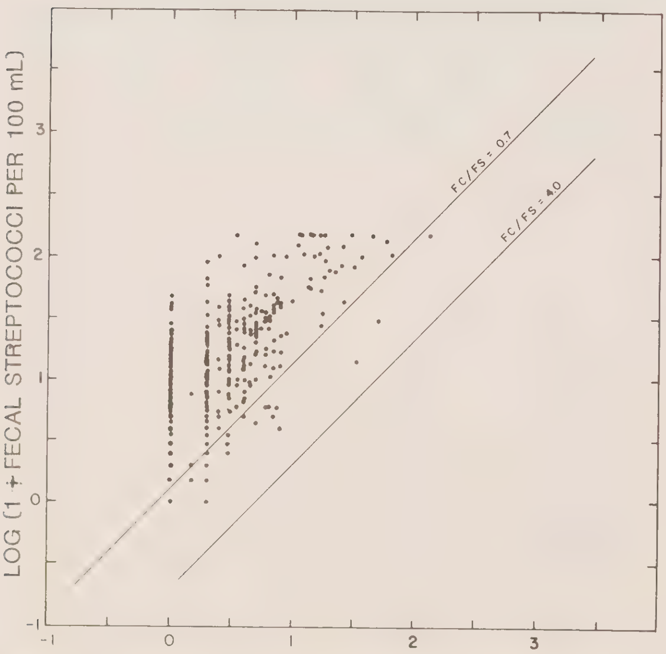


Figure 11: Plot of annual median levels of fecal coliforms versus fecal streptococcus (on logarithmic scales) for all locations on all lakes for the years 1976 to 1979. Each point is from one location on a lake for a year. The lines for fecal coliform versus fecal streptococcus ratios of 4.0 and 0.7 have been added.

Again, no specific relationship was detected between fecal coliform and fecal streptococcus levels. However, an increase in the fecal coliform levels did correspond to an increase in the fecal streptococcus levels.

In the literature, when fecal streptococcus levels are in the range of 100 per 100 mL or greater and the data represents the water quality near a source of pollution, the fecal coliform to fecal streptococcus ratio suggests the type of contamination from that source. In Figure 11, the lines corresponding to the two critical points of the fecal coliform to fecal streptococcus ratio have been drawn. On the logarithmic scale, these ratios are parallel lines with slope one and intercepts of 0.15 and −0.60 for non-human and human resources respectively. All points to the right of the human source ratio line would give a ratio greater than 4 and therefore would be expected to be from a human fecal source. All points to the left of the non-human source ratio line would give a ratio less than 0.7 and therefore would be expected to be from a non-human source. The majority of the data points were to the left of the 0.7 ratio line and most fecal streptococcus levels were below 100 per 100 mL.

In addition to the pollution indicators, samples from mid-lake locations were analyzed for heterotrophic bacteria. This procedure enumerates all bacteria requiring organic carbon as an energy source and capable of growing on a minimal low nutrient medium at a temperature of 20°C. Early work within the Lakeshore Capacity Study suggested that the levels of heterotrophic bacteria might be related to ''nutrient'' levels or trophic status (Hendry 1977). This conclusion is a logical extension of standard microbial ecology but further investigation of this relationship was discontinued because of other workloads within the study. However, analysis for heterotrophic bacteria on a restricted number of samples was continued. The annual median levels of heterotrophic bacteria ranged from 300 to 300,000 bacteria per mL. However, the levels greater than 10,000 heterotrophs per mL represent lakes sampled in 1977. The lakes sampled in 1978 had levels in the range 3000 to 10,000 heterotrophs per mL. This variation may have represented an actual yearly variation in heterotroph levels, but more likely reflected a lake-to-lake variation since few of the lakes examined in 1977 were also examined in 1978 or 1979 (Table 1).

4.1.2 BEACHES AND BACTERIAL INDICATORS

At the beaches total coliform levels tended to be higher than those found in the rest of the lake. Median total coliform levels ranged from 10 to 10,000 per 100 mL. The majority of these levels compared favourably with the Ontario objective for recreational use. However, some did exceed the 1000 total coliform per 100 mL.

The fecal coliform levels at the beaches also tended to be higher than those found in the rest of the lake. However, most levels did compare favourably with the Ontario recreational use objective. Fecal coliform levels ranged from 0 to 100 per 100 mL. The relationship between the total coliform and fecal coliform levels at the study beaches is presented in Figure 12. As was seen in the lake sampling, no specific relationship was detected between total coliform and fecal coliform levels.

The median fecal streptococcus levels found at the study beaches ranged from 0 to 500 per 100 mL with the majority of levels between 10 and 100 per 100 mL. These levels tended to

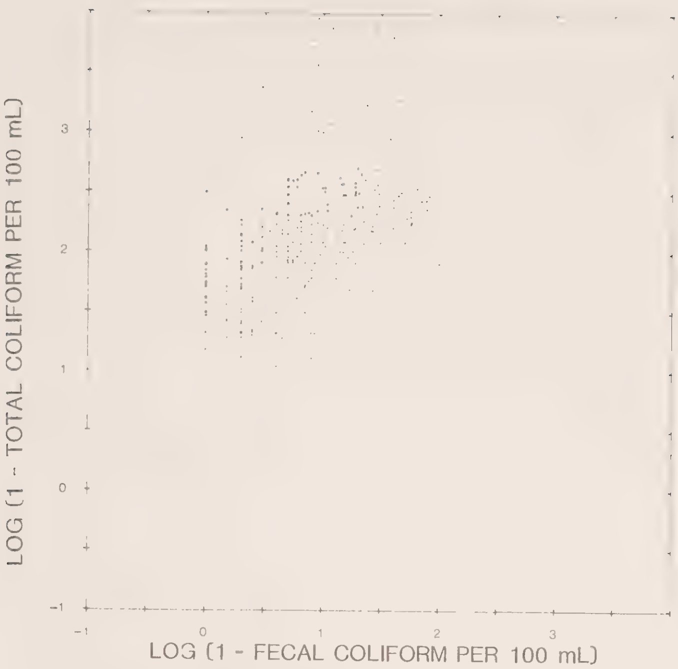


Figure 12: Plot of median levels of total coliforms versus fecal coliforms (on logarithmic scales) for all sectors of the beaches and all observation times for 1979.

be slightly higher than corresponding levels at other locations on the lake. Figure 13 presents the relationship between fecal coliform and fecal streptococcus levels as found at the study beaches. The fecal streptococcus levels generally were higher than the fecal coliform levels suggesting a non-human source of pollution.

4.1.3 OCCURRENCE OF *PSEUDOMONAS AERUGINOSA* — LAKES AND BEACHES

The bacterium *P. aeruginosa* has been described as "a ubiquitous organism, able to flourish in multiple environments, ...to survive for lengthy periods as long as sufficient moisture is available" (Young 1977). In the study lakes, *P. aeruginosa* was found to be far from ubiquitous (Table 6) occurring in, at most, 20% of the lake samples. The apparent increase from 1976 to 1979 probably reflected a shift in sampling sites, in order to study locations where *Pseudomonas* played a more significant role.

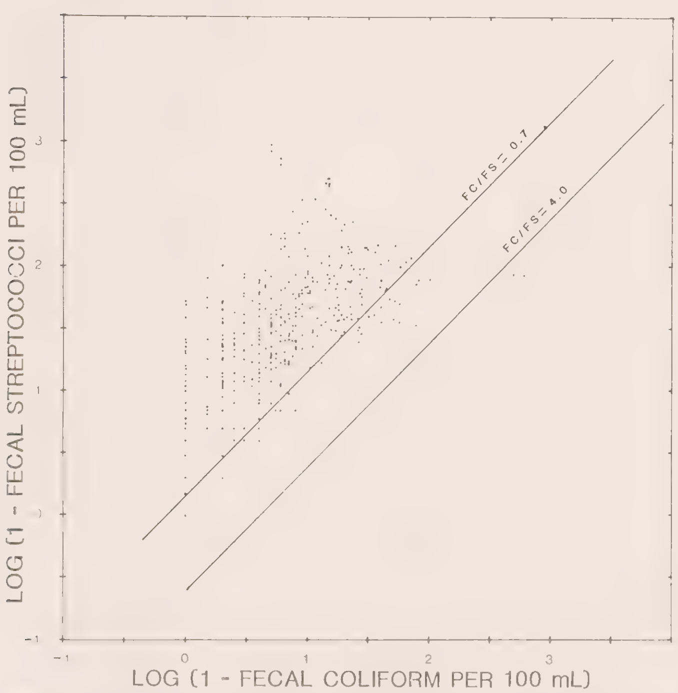


Figure 13: Plot of median levels of fecal coliform versus fecal streptococcus for all sectors of the beaches and all observation times for 1979. The lines corresponding to the fecal coliform to fecal streptococcus ratios of 4.0 and 0.7 have been added.

Table 6. Summary of the occurrence of *P. aeruginosa* in lake water samples from 24 Lakeshore Capacity Study lakes during the period 1976 to 1979.

Year	Total Number of Samples	Number of Samples Containing *P. aeruginosa* # (%)
1976	257	6 (2%)
1977	725	41 (6%)
1978	914	107 (12%)
1979	1084	217 (20%)

The levels of *P. aeruginosa* are presented in Figures 14, 15 and 16. *Pseudomonas* was not detected in the majority of samples at most locations. Occasionally samples had extremely high levels of *Pseudomonas* (e.g. 100 to 500 *Pseudomonas* per 100 mL). Seasonal variation was apparent with the summer period having the highest *Pseudomonas* levels. However, there were occasional exceptions to that general seasonal trend. Mid-lake locations (Figure 15) had higher levels of *Pseudomonas* than shoreline (Figure 14) or inflow/outflow (Figure 16) locations. Two factors affect this observation: 1) Each bar on Figure 15 generally summarizes four to eight samples, while the corresponding bar on Figure 14 summarizes twenty to eighty samples. Therefore, Figure 15 is more prone to reflect extreme case conditions. 2) At the time of analysis, the technicians noted the unusually high *Pseudomonas* levels at a mid-lake location but also noted that these high levels usually occurred shortly after or during a rainstorm. This would suggest a washing in of bacteria from an external source.

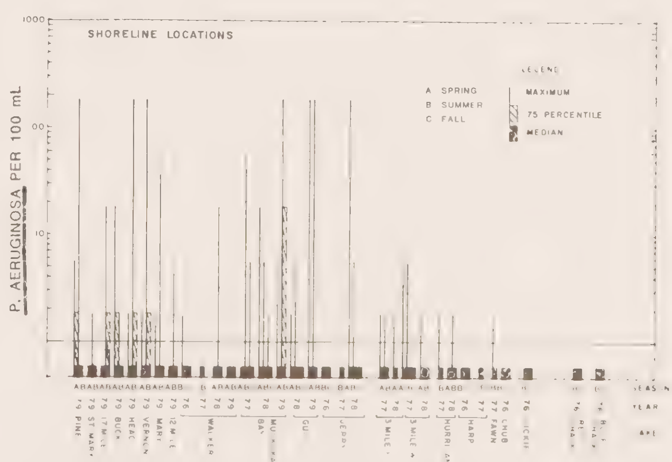
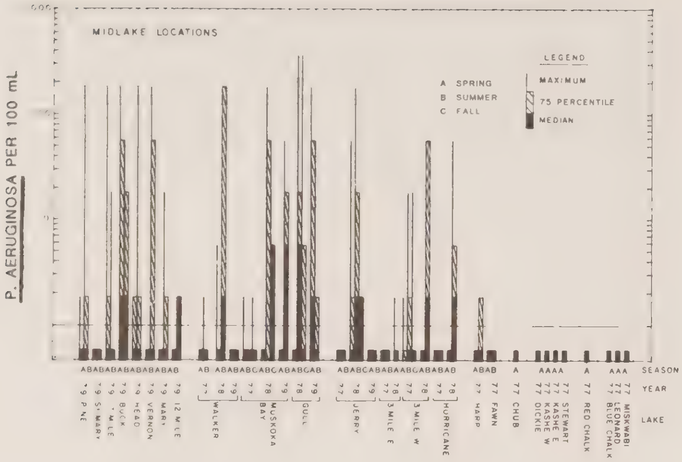


Figure 14. Plot of seasonal median, 75 percentile, and maximum levels of *P. aeruginosa* per 100 mL for all shoreline locations by lake, season and year. The seasons used are: spring (A) - May and June; summer (B) - July and August; and fall (C) - September to November.



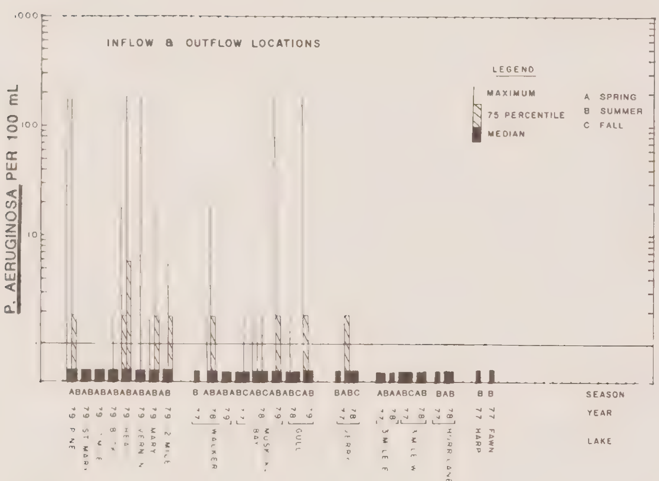


Figure 16: Plot of seasonal median, 75 percentile, and maximum levels of *P. aeruginosa* per 100 mL for all inflow/outflow locations by lake, season and year. The seasons used are: spring (A) – May and June; summer (B) – July and August and fall (C) – September to November.

P. aeruginosa was isolated from between 21 and 41 percent of the samples taken at the study beaches during 1979 (Table 7). These percentages were much higher than the 1 to 20 percent figures seen for lake shoreline (Table 6). But the majority of samples still did not contain this organism.

The levels of *Pseudomonas* were also dependent on the time of year (Figure 17). Both the frequency of isolation and the levels of *Pseudomonas* were higher in August of 1979 than in either June or July. *Pseudomonas* were usually absent during other times of the year. Single samples may have had up to 1000 *P. aeruginosa* per 100 mL.

Table 7. Numbers of samples and isolation frequency of *P. aeruginosa* at the 1979 study beaches.

Beach (1979)	Total Number of Samples	Number of Samples Containing *P. aeruginosa* # (%)
Gull	146	39 (27%)
Head	128	53 (41%)
Mary	152	41 (27%)
Pine	59	23 (39%)
Vernon	174	49 (28%)
Twelve Mile Public	64	15 (23%)
Twelve Mile Red Cross	63	13 (21%)

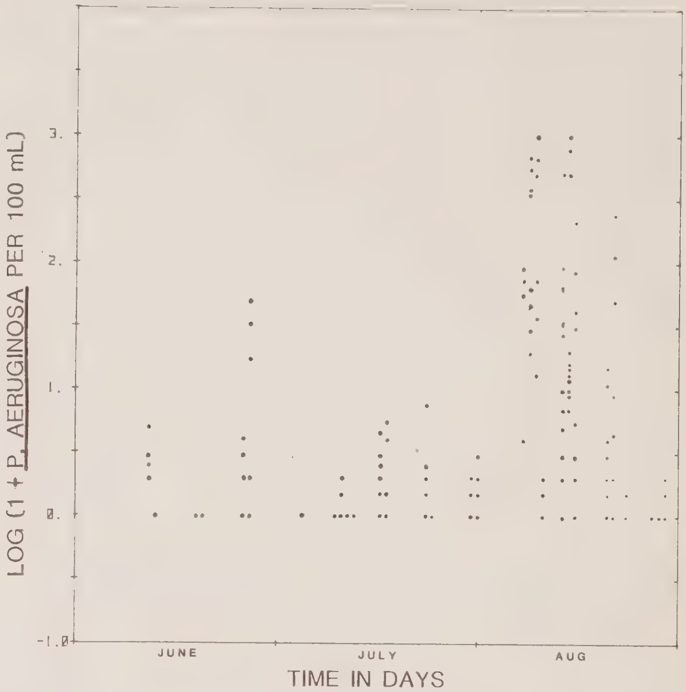


Figure 17: Time trend of *P. aeruginosa* levels observed for all sectors of the 1979 study beaches.

4.1.4 DEVELOPED VERSUS UNDEVELOPED SHORELINE

The bacterial water quality of developed shorelines with human activity, cottages, etc. were compared with that of natural shorelines with only trees, rocks and animals on the same lake or similar lakes. An undeveloped lake, Jerry Lake, has been under study for a number of years and its water quality compared with other nearby lakes. In general, the water quality at the undeveloped and the developed shorelines behaved in a similar fashion. Differences in microbiological water quality were marginal or nonexistent. Public access beaches seemed to have enough intensity of use for an impact to be measurable by the current microbiological procedures.

4.2 CHEMICAL WATER QUALITY

Table 8 summarizes in annual medium form the results of the chemical examination from the mid-lake and beach sampling locations studied in 1979. As with the microbial water quality, the chemical results did show variations from lake to lake, within the same lake at different locations and across time. Walker, Twelve Mile and Buck Lakes showed lower levels of dissolved Kjeldahl nitrogen. The difference between total and dissolved Kjeldahl nitrogen levels in Head and Buck Lakes and Muskoka Bay suggested a significant contribution from particulate material. Higher levels of nitrate and nitrite in Mary Lake, Vernon Lake and Muskoka Bay suggested a significant difference in the type of input to these lakes or possibly a greater conversion of other forms of nitrogen to nitrate by the resident bacterial population. Head and Pine Lakes had higher levels of inorganic carbon suggesting a "harder" water. Conductivity was low and only slightly higher than that of precipitation in the area. Lakes would be categorized as oligotrophic or mesotrophic.

Table 8. Water quality at mid-lake and beach locations during the 1979 sampling season (May through October). At mid-lake the median summarizes the data from two locations. At beaches, the median summarizes the data from two locations at two times on each sampling day. The number of values summarized is given at the bottom of the table.

Test	Units	Median Chemical Concentrations				
		Buck Lake	Gull Lake	Gull L. Beach	Head Lake	Head L. Beach
total Kjeldahl nitrogen	ug/L	234	345	384	440	410
dissolved Kjeldahl nitrogen	ug/L	236	350	370	365	365
nitrite plus nitrate	ug/L	5	5	5	5	5
ammonium	ug/L	12	7	5.4	1.8	19.5
total phosphorus	ug/L	6.8	8.1	8.1	13.5	9.7
total carbon	mg/L	5.6	5.8	6.6	10.6	10.3
total organic carbon	mg/L	3.8	7.8	4.7	5.7	5.2
inorganic carbon	mg/L	1.7	2.0	1.8	5.1	5.1
pH	—	6.4	6.7	6.7	7.1	7.1
conductivity	micromhos/cm	34	59	59.8	74	75
water temperature	°C	20.2	20.8	21.0	21.7	22.4
dissolved oxygen	mg/L	8.8	8.6	—	8.4	—
secchi disk	m	7.8	3.4	—	3.5	—
colour	C.U.	5	5	5	12	5
chlorophyll A	ug/L	1.4	2.0	—	3.4	—
Number of values		10	22	32	18	28

— not available

Test	Units	Median Chemical Concentrations				
		Mary Lake	Mary L. Beach	Muskoka Bay	Pine Lake	Pine L. Beach
total Kjeldahl nitrogen	ug/L	329	322.5	382	312	331.3
dissolved Kjeldahl nitrogen	ug/L	316	305	320	327	287
nitrite plus nitrate	ug/L	188	185	172	17	10
ammonium	ug/L	14	15	12	11	4.1
total phosphorus	ug/L	7.4	7.4	12.1	8.8	8.7
total carbon	mg/L	6.7	6.2	6.8	7.6	7.4
total organic carbon	mg/L	5.5	5.2	4.6	4.2	4.4
inorganic carbon	mg/L	1.0	1.0	2.2	3.5	3.1
pH	—	6.5	6.6	6.9	7.0	6.9
conductivity	micromhos/cm	43	43.5	62	57	55.5
water temperature	°C	20.0	20.8	18.6	20.0	20.8
dissolved oxygen	mg/L	8.8	—	8.6	8.6	—
secchi disk	m	3.8	—	4.0	5.2	—
colour	C.U.	10	10	5	5	5
chlorophyll A	ug/L	1.8	—	37	2.0	—
Number of values		24	34	14	12	16

— not available

Test	Units	Median Chemical Concentrations				
		St. Mary Lake	17 Mile Lake	12 Mile Lake	12 Mile L. (Public)	12 Mile L. (Red Cross)
total Kjeldahl nitrogen	ug/L	268	410	237	240	282.5
dissolved Kjeldahl nitrogen	ug/L	265	265	225	215	276.3
nitrite plus nitrate	ug/L	46	15	50	32.5	10
ammonium	ug/L	14	11	10	8.5	8.5
total phosphorus	ug/L	9.4	8.6	5.2	6.4	6.1
total carbon	mg/L	6.6	5.8	6.4	6.2	5.9
total organic carbon	mg/L	5.0	3.8	3.8	3.5	3.5
inorganic carbon	mg/L	1.9	1.6	2.4	2.5	2.4
pH	—	6.7	6.4	7.1	7.0	6.9
conductivity	micromhos/cm	45	36	52	52	51
water temperature	°C	19.0	20.6	19.4	19.7	22.0
dissolved oxygen	mg/L	9.1	8.4	9.2	—	—
secchi disk	m	3.8	6.8	6.0	—	—
colour	C.U.	5	5	5	5	5
chlorophyll A	ug/L	2.2	2.0	1.6	—	—
Number of values		8	10	8	20	11

— not available

Test	Units	Median Chemical Concentrations: Vernon Lake	Vernon L. Beach	Walker Lake
total Kjeldahl nitrogen	ug/L	342	332.5	245
dissolved Kjeldahl nitrogen	ug/L	317	325	218
nitrite plus nitrate	ug/L	130	115	18
ammonium	ug/L	11	11	12
total phosphorus	ug/L	7.6	8.7	7.6
total carbon	mg/L	6.7	6.6	5.1
total organic carbon	mg/L	5.6	5.6	3.7
inorganic carbon	mg/L	0.9	0.9	1.3
pH	—	6.4	6.5	6.6
conductivity	micromhos/cm	38	38	36
water temperature	°C	18.8	20.0	18.0
dissolved oxygen	mg/L	8.5	—	9.2
secchi disk	m	3.0	—	5.5
colour	C.U.	15	15	5
chlorophyll A	ug/L	23	—	2.4
Number of values		24	36	10

Some variation in chemical levels can be seen between the corresponding beach and lake locations but, in general, they are the same.

4.3 EAR INFECTIONS AND HUMAN POPULATIONS

4.3.1 INTRODUCTION

Throughout the study, swimming activity was considered the same at different locations if the same number of people were using the swimming facility. The origin of the people did not affect the swimming activity. For example, the swimming activity which occurred in front of a single cottage was similar to the swimming activity which occurred at a beach available to the general public. In front of a single cottage, the swimming activity would be zero to ten people per swimming area. At a public beach the total number of people using the swimming area may be considerably higher (Figure 18). The number of people actually in the water varies with the time of day and the day of the year. More people used the swimming area from mid-July to mid-August and between the hours of 10 AM and 3 PM. A weekly pattern also existed but was easily altered by climatic conditions.

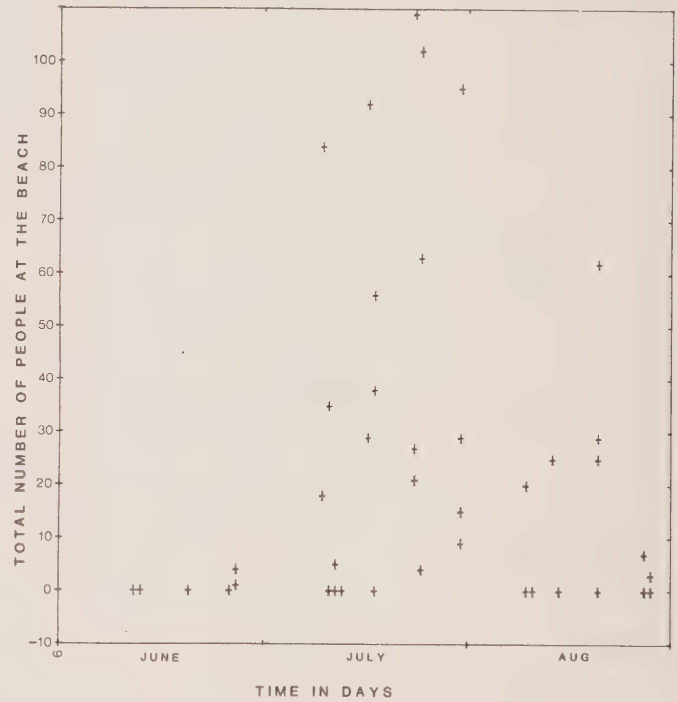


Figure 18: Total number of people at a study beach at any particular time related to the time of year during the summer of 1979. All beach studies are represented in this graph.

Table 9. Age distribution of respondents to the 1977, 1978 and 1979 questionnaires. Column labelled "Ear Infection" show age distribution of those reporting ear infections. "General Response" shows the age distribution across a number of study locations for that year.

Age Range (years)	1977*: Muskoka Bay	1977*: Gull Lake Beach	1978: General Response	1978: (%)	1978: Ear Infection	1979: General Response	1979: (%)	1979: Ear Infection
0-4	4%	0%	1%	-	-	9%	(13%)	16%
5-14	13%	93%	47%	(15%)	71%	40%	(12%)	61%
15-24	21%	7%	17%	(8%)	13%	17%	(4%)	10%
25-34	16%	-	9%	(9%)	8%	14%	(4%)	8%
35-44	11%	-	11%	(2%)	3%	8%	(1%)	1%
45-54	12%	-	5%	(10%)	5%	4%	(2%)	1%
55-64	13%	-	4%	-	-	2%	(4%)	1%
65-74	6%	-	4%	-	-	3%	(3%)	1%
75-84	1%	-	1%	-	-	1%	-	-
No answer	4%	-	1%	-	-	-	-	-
Total number of individuals	531	55	380		38	1042		79

(%) = the percentage of the respondents in that category which also reported an ear infection.
- = not applicable.
* = No ear infections reported in 1977.

By using these two situations, the cottage and beach, the study was examining swimming activities where the swimmer density varied from very low to moderate use.

Table 10. Summary of response to the 1977 Muskoka Bay questionnaire.

Question	Yes %	No %
Male	50	—
Female	50	—
Frequency of water contact		
less than 3 times per week	33	—
3-5 times per week	22	—
more than 5 times per week	29	—
did not swim	12	—
Swimming location		
cottage	74	—
other locations on lake	8	—
pools	2	—
others	8	—
Wear ear plugs or bathing cap	—	80
ear plugs	1	—
bathing cap	5	—
Wet head while swimming	74	11
Dives	51	31
Predisposition to ear infection	7	81
Total number of questionnaires	531	

— not applicable

4.3.2 1977 MUSKOKA BAY COTTAGER SURVEY

In 1977, 136 cottages were visited on Muskoka Bay and 531 people were interviewed. Of these individuals 264 were male and 265 were female. The age distribution of the people responding is given in Table 9.

The majority of people interviewed at Muskoka Bay in 1977, swam less than three times per week, and only at their cottage. They did not wear ear plugs or a bathing cap, they wet their heads while swimming, dived and they believed they were not predisposed to an ear infection (Table 10).

There were no ear infections reported in the summer of 1977 at Muskoka Bay.

Table 11. Summary of response to the 1977 Gull Lake questionnaire.

Question	Response (%) Yes	No
Male	47	—
Female	53	—
Wearing a bathing cap	0	100
Wear ear plugs	2	98
Head wet while swimming	100	0
Dive	100	0
Taking medication	33	67
Specific type of medication		
Ear infection medication	20	—
Allergy medication	6	—
Not specified	7	—
Swim at other locations frequently	76	24
Total number of questionnaires	55	

— not applicable.

4.3.3 1977 GULL LAKE SWIMMING CLASS SURVEY

In the survey conducted at Gull Lake in 1977, 55 people were interviewed. There were 26 males and 29 females. The majority of those interviewed were in the range of 5 to 14 years (Table 9). All of the people interviewed were members of a swimming class.

The majority of the people interviewed did not wear bathing caps or ear plugs, got their head wet while they swam, dived, were not taking medication of any kind and generally swam at other locations on a frequent basis (Table 11).

There were no cases of *otitis externa* reported by the people interviewed at Gull Lake.

Table 12. Location of questionnaire interviews during 1978 and 1979.

Location	Percent of Questionnaires 1978	1979
Gull Lake Beach	47	26
Muskoka Bay	27	—
Mouse Lake	5	2
Bass Lake	4	—
Peninsula Lake	1	—
Lake of Bays	2	—
Vernon Lake Beach	2	15
Mary Lake Beach	2	19
Head Lake Beach	8	9
Pine Lake Beach	—	12
Twelve Mile Lake Beach Red Cross	—	15
Twelve Mile Lake Beach Public	—	1
Unclassified	2	—
Total number of questionnaires	380	1042

— not applicable.

Table 13. Summary of response to the 1978 swimmer questionnaire.

Question	General Response Yes		No		Response of those reporting an ear infection Yes	No
Male	44%	(8%)	—		34%	—
Female	56%	(12%)	—		66%	—
Dive	71%	(11%)	24%	(8%)	79%	18%
Swim under water	82%	(11%)	13%	(4%)	92%	5%
Wear a bathing cap	5%	(10%)	90%	(10%)	5%	92%
Wet head while swimming	86%	(11%)	9%	(3%)	95%	3%
Wear ear plugs	2%	(13%)	93%	(10%)	3%	95%
Previous infection	37%	(26%)	59%	(1%)	95%	5%
Often ear infection	4%	(44%)	91%	(9%)	18%	82%
Infection this summer	10%		—		—	—
Associate infection this summer with swimming	7%	(93%)	16%	(16%)	68%	26%
Seek medical treatment	6%	(90%)	18%	(27%)	50%	47%
Ever hospitalized	2%	(11%)	40%	(24%)	3%	95%
Medication prior to symptoms of ear infection	—	—	25%	(39%)	—	97%
Total number of questionnaires	380				38	

(%) the percentage of the respondents in that category which also reported an ear infection.

— not applicable.

4.3.4 1978 SWIMMER QUESTIONNAIRE

In 1978, 380 people were interviewed at various locations (Table 12). Of these, 214 (56%) were female and 166 (44%) were male (Table 13). Sixty-five percent of the people interviewed were 24 years old or younger (Table 9).

The majority of the general population did not swim in May or June. However, the frequency of swimming did increase in July and August. A large portion of the people said they swam 10 or more hours per week in July and August (Table 14).

Table 14: Distribution of ear infections and swimming frequency during May, June, July and August of 1978.

Swimming Frequency Response	May			June			July			August		
	General		% ear	General		% ear	General		% ear	General		% ear
Did not swim	69%	(10%)	66%	35%	(8%)	29%	4%	-		3%	-	
Occasionally	19%	(13%)	24%	28%	(8%)	21%	16%	(2%)	3%	17%	(2%)	3%
Frequently	9%	(9%)	8%	26%	(14%)	37%	38%	(8%)	31%	39%	(8%)	31%
Very Frequently	1%	(20%)	2%	9%	(15%)	13%	40%	(16%)	66%	39%	(17%)	66%
No answer	2%	-		2%	-		2%	-		2%	-	

Occasionally 0-2 hrs./wk., Frequently 3-10 hrs./wk., Very frequently 10 or more hrs./wk.
(%) = the percentage of the respondents in that category which also reported an ear infection.
- = not applicable
% ear = response of those reporting an ear infection.

Of the 380 people interviewed, most of them said they dived, swam underwater, got their heads wet while swimming and did not wear a bathing cap or ear plugs (Table 13). One hundred nineteen people (37%) had a previous ear infection and 16 people (4%) reported that they often experienced ear infections. The majority of the general population had never been hospitalized for an ear infection at any time.

In 1978, 38 people (10%) of the total 380 people interviewed reported having an ear infection during the summer. Of these, 25 (66%) were female and 13 (34%) were male. Seventy-one percent of the people reporting ear infections were 5 to 14 years old (Table 9). Of the people reporting ear infections, the majority dived, swam underwater, got their head wet while swimming and did not wear a bathing cap or ear plugs.

Thirty-six (95%) of the 38 people reporting ear infections had previously experienced ear infections but this majority also reported that they did not often experience ear infections. Sixty-eight percent of the people did not associate their ear infection with their swimming activity, and most of the people sought medical treatment for their ear infection in the summer of 1978. A large number of the people with ear infections were never hospitalized for these and were not taking medication for another ailment at the time the symptoms of the ear infection appeared.

The majority of the population experiencing ear infections, did not swim in May, swam frequently (3 to 10 hours per week) in June and swam very frequently (10 or more hours per week) in July and August (Table 14). The highest percentage of the people with ear infections in the summer of 1978 experienced them in August (42%). Twenty-six percent of the ear infections occurred in July.

4.3.5 1979 SWIMMER QUESTIONNAIRE

In 1979, a total of 1042 people were interviewed at 8 different locations (Table 12). Six hundred forty (61%) were female and 402 (39%) were male (Table 15). More than half of the people interviewed were 24 years or younger (Table 9).

Table 15. Summary of response to the 1979 swimmer questionnaire.

Question	General Response				Response of those reporting an ear infection	
	Yes		No		Yes	No
Male	39%	(8%)	—		42%	—
Female	61%	(7%)	—		58%	—
Head wet while swimming	87%	(8%)	12%	(3%)	96%	4%
Swimming class	32%	(13%)	67%	(5%)	56%	44%
Untreated water for washing	19%	(4%)	44%	(13%)	20%	33%
Previous ear infection	26%	(27%)	74%	(1%)	90%	10%
Ever treated for an ear infection	22%	(26%)	72%	(2%)	79%	20%
Ever been hospitalized for an ear infection	2%	(44%)	91%	(7%)	10%	81%
Ever treated yourself for an ear infection	6%	(28%)	87%	(6%)	22%	71%
Ear infection this summer	8%		—		—	—
Treated this summer	6%	(94%)	87%	(2%)	73%	22%
Associate with swimming activity	11%	(38%)	75%	(4%)	54%	38%
Regularly swim at more than one beach	66%	(8%)	32%	(46%)	68%	32%
Total number of questionnaires	1042				79	

(%) the percentage of the respondents in that category which also reported an ear infection.

— not applicable.

The majority of the general population swam very frequently (4 or more times per week), got their heads wet while swimming, were not enrolled in a swimming class and did not use untreated water for washing or bathing (Tables 15, 16). Most of the general population did not experience a previous ear infection, were never treated by a doctor or hospitalized for an ear infection, never tried to treat themselves for an ear

infection, and said they were not treated for an ear infection that summer. Most of the people interviewed swam at more than one beach.

In 1979, 79 people (8%) of the 1042 interviewed reported experiencing an ear infection. Of these, 46 (58%) were female and 33 (42%) were male (Table 15). Seventy-five percent of the people experiencing an ear infection were 14 years or younger (Table 9). Most of the people experiencing an ear infection swam very frequently (4 or more times per week), got their head wet while swimming and were members of a swimming class. Seventy-one (90%) of those reporting an ear infection had experienced a previous ear infection. The majority of the infected population were treated at some time for a previous ear infection. Most of the people had never been hospitalized and had never tried to treat themselves for an ear infection.

Table 16. The frequency of swimming activity of the general population and those reporting an ear infection in 1979.

Frequency of swimming	Response: General Population		Those Reporting Ear Infection in 1979
Did not swim	5%	(2%)	1%
Occasionally	20%	(2%)	5%
Frequently	18%	(7%)	17%
Very frequently	56%	(10%)	77%

Occasionally = 1-2 ×/week, less than 1 hour
Frequently = 1-4 ×/week
Very frequently = 4 or more times/week
(%) = the percentage of the respondants in that category which also reported an ear infection.

Of the people experiencing an ear infection in the summer of 1979 the majority were treated for their ear infection. Most of the people associated their ear infection with their swimming activity and most swam regularly at more than one beach.

Forty-six percent of the ear infections occurring in the summer of 1979 were in the month of July, eighteen percent in June and a considerably smaller number (6%) occurred in August.

4.3.6 THE *OTITIS EXTERNA* PATIENT SURVEYS OF 1978 AND 1979

In 1978, a total of 79 cases of *otitis externa* were reported by participating physicians in 4 communities (Table 17). In 1979, a total of 114 cases were reported by physicians in 5 communities.

Table 17. Summary of communities responding to patient questionnaires in 1978 and 1979.

Location	Patients Only 1978	Patients Only 1979	Physician Only 1978	Physician Only 1979	Complete data* 1978	Complete data* 1979	Total 1978	Total 1979
Bracebridge	4	1	3	0	1	5	8	6
Bracebridge Emergency	0	2	6	3	14	10	20	15
Huntsville Emergency	3	1	5	1	7	20	15	22
Huntsville	0	1	1	1	2	13	3	15
Haliburton	0	0	1	4	28	17	29	21
Gravenhurst	0	1	3	0	1	0	4	1
Minden	-	2	-	7	-	25	-	34
Total	7	8	19	16	53	90	79	114

- not applicable
* Both Patient and Physician responses

Table 18 indicates the age distribution of the patients in 1978 and 1979. The majority of the patients experiencing an ear infection were between the ages of 5 and 24 years.

Table 18. Summary of age distribution for 1978 and 1979 *otitis externa* patient surveys.

Age Group (years)	Number of People (%) 1978	1979
0-4	3	1
5-14	47	64
15-24	30	22
25-34	7	3
35-44	8	8
45-54	3	0
55-64	0	1
65-74	0	0
75-84	2	0
NA	0	2
Total number	60	98

Table 19 presents the general response to the questions on both the 1978 and 1979 patient questionnaires. In 1979, 97 people responded to the patient questionnaire and in 1978, 60 people responded. In both years male patients outnumbered female patients. The majority of the patients in both years dived, swam underwater, wet their head while swimming and did not wear a bathing cap or ear plugs. Almost half the people responding in 1978 and 1979 said thay had experienced a previous infection and the majority of the patients associated their ear infection with their swimming activity.

Table 19. Summary of response to the *otitis externa* patient questionnaire of 1978 and 1979.

Response	1978 Yes (%)	1978 No (%)	1979 Yes (%)	1979 No (%)
Male	57	—	59	—
Female	43	—	41	—
Dive	80	17	77	21
Swim underwater	92	5	95	3
Bathing cap	0	97	1	97
Wet head	93	2	97	1
Ear plugs	2	92	3	95
Previous infection	50	45	51	47
Associate infection	75	17	85	13
Hospitalized	—	—	5	93
Ever treated	—	—	22	76
Treated without medical advice	—	—	13	85
Untreated water	—	—	51	47
Swimmer	—	—	99	1
Often swimming:				
per week 1-7 x.	62	—	—	—
8-15 x	8	—	—	—
+16 x	25	—	—	—
Length of time in water per swim:				
1/4-1 hr.	43	—	—	—
1-3 hrs.	53	—	—	—
Total number of questionnaires	60		97	

Most of the patients in 1979 had never been hospitalized for an ear infection and had never treated themselves for an ear infection with medication prescribed for a previous ear infection. Eighty-five percent of the patients in 1979 indicated they had never tried to treat themselves for an outer ear

infection without obtaining medical advice. About one-half of the patients indicated that they used untreated lake or river water for washing or bathing.

4.4 RELATIONSHIP BETWEEN *P. AERUGINOSA* AND OTHER VARIABLES

In the *P. aeruginosa* model, certain variables were suggested to have some relationship with the levels of *Pseudomonas* in water. These variables were suggested because the literature indicated that a relationship existed or because general microbiology principles indicated that a relationship could exist. The prime variables were: 1) some measure of human user density, e.g. total people at the beach, because this would be the source of the bacteria; 2) the water temperature, because growth/survival of the bacteria was favoured by warm temperatures and 3) some measure of nutrients, because these would be needed by the bacteria if they were to grow.

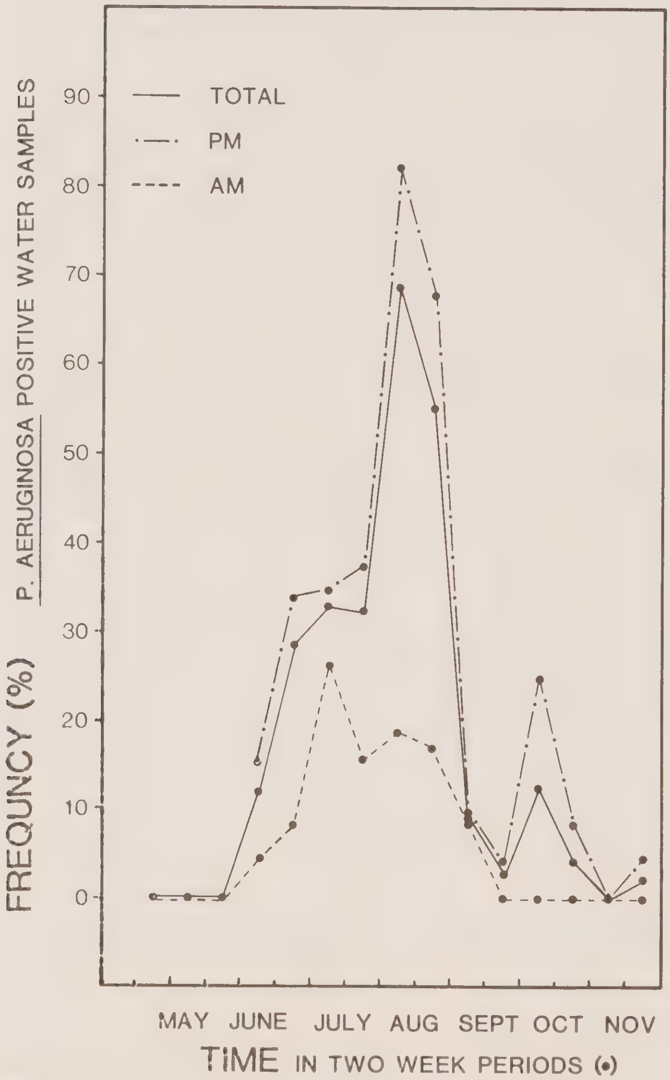


Figure 19: Time trend of the frequency of isolation of *P. aeruginosa* from water samples during 1978 at Gull Lake beach.

For each two week period during 1978, the water samples were categorized as samples taken prior to the daily use of the beach, usually in the morning (AM) or samples taken after the beach had been used (PM).

The third category (TOTAL) contained all samples. Each line summarizes the percentage of samples in the category which contained *Pseudomonas*.

The results from the 1978 Gull Lake beach study seemed to confirm the preconceived relationships in a qualitative way. The frequency of isolation of *P. aeruginosa* from the water at the beach followed a peaked time trend (Figure 19) with maxium values attained in late July and early August rising in early June and declining in early September. The numbers of users (Figure 20) of the beach followed a similar time trend to the *Pseudomonas* frequency of isolation and also showed a similar diurnal variation since *Pseudomonas* was more frequently isolated in the afternoon than the morning. This time trend was also repeated in water temperatures (Figure 21). Therefore, at least two variables, number of beach users and water temperature, could be related to *P. aeruginosa* levels.

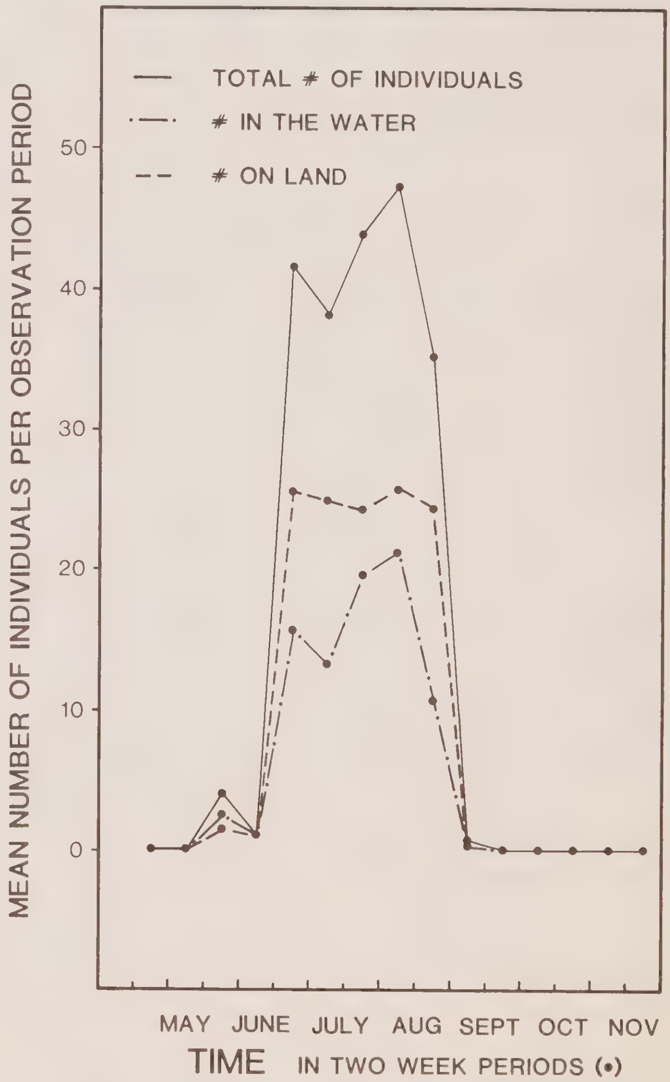


Figure 20: Time trend of the use of Gull Lake beach during 1978. The lines represent the total number of individuals at the beach, the numbers who were physically in the water at the time the tally was taken and the number on the sand part of the beach. The mean number for each two week period is given but the extremes are in the range 0 to 120 individuals.

The Gull Lake beach results were then examined statistically utilizing standard transformations of the data and a stepwise multiple linear regression. The multiple regression procedure revealed (equation A on Table 20) that the logarithm of the *Pseudomonas* level was related to the water temperature, the total number of users of the beach in the previous hour and the logarithm of the fecal coliform level. The variable, fecal

coliform, was the least significant variable in the equation but did fit the inverse relationship suggested in the literature (Mucha et al 1969).

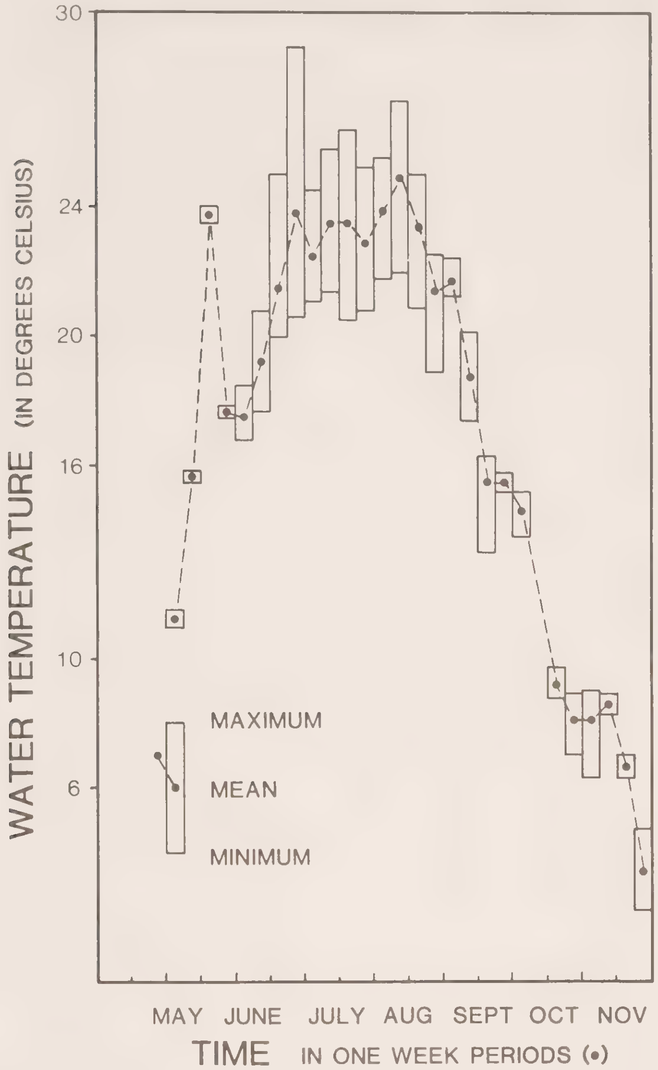


Figure 21: Time trend of the water temperature at Gull Lake beach during 1978. Mean maximum and minimum water temperatures for each one week period are shown.

The 1979 results followed similar time trends to the 1978 results. Some of the beaches showed a secondary maximum in frequency of *Pseudomonas* isolations in July. However, using the multiple regression procedure, the equations B through F (Table 20) were derived. No two of the equations showed the same significant variables. Many of the relationships were hard, if not possible, to explain. The relationships did not fit a model.

Examination of some of the underlying statistics did provide some answers to the non-uniformity of multiple linear regression relationships. The multiple linear regression assumed that all variables were either normally distributed or were transformed to be so. The logarithmic transformation is commonly assumed to normalize microbiological data. However, the low level bacterial densities present in this study were not distributed according to a log normal distribution but were distributed according to a truncated log normal distribution. The truncated log normal distribution arose because all results in the lower tail of the distribution are reported as no bacteria detected in a given volume of sample. No *P. aeruginosa* was detected in up to 90% of 100 mL samples taken from specific locations. Similar statements can be made for fecal coliform and in some cases fecal streptococcus levels.

All of the evidence from the Lakeshore Capacity studies did suggest that a relationship existed between *P. aeruginosa*, the number of people using the beach, and the water temperature but the relationship could not be statistically validated. *P. aeruginosa* tended to increase as the numbers of users of the beach increased. *Pseudomonas* could not usually be isolated where the users from single cottages utilize a shoreline but could usually be isolated at heavily used beaches. The relationship fit the literature statement that *Pseudomonas* can be isolated in "northern temperature surface waters" only if the water was "recently affected by human activity or the activities of domestic animals" (Hoadley 1977).

Increasing water temperature seemed to increase the levels of *Pseudomonas* by providing a mechanism for the bacterium to either survive for a longer period of time or to grow. This relationship is also suggested in the literature (Hoadley 1977).

Table 20. Relationship of *P. aeruginosa* with other variables measured in the water and at the beach. All of the equations were developed using the stepwise multiple linear regression procedure until all remaining variables were not significantly related to the developing equation.

Location	Equation	Statistics		
		r^2	F	d.f.
Gull Lake Beach 1978	A log (1 + PsA) = − 0.72 + 0.041 WTEMP − 0.164 log (1 + FC) + 0.0048 TPREV	0.24	10.9	3, 100
Gull Lake Beach 1979	B log (1 + PsA) = − 2.36 + 0.0047 BPREV − 0.0005 DTKN + 0.044 COND	0.33	6.61	3, 39
Pine Lake Beach	C log (1 + PsA) = − 1.415 + 1.044 log (1 + TC) + 0.022 TPREV	0.92	77.3	2, 13
Vernon Lake Beach	D log (1 + PsA) = − 49.27 − 0.21 DAY − 0.031 BPREV + 0.251 TP − 0.008 DTKN + 0.616 pH − 6.87 COND	0.93	39.8	6, 19
Head Lake Beach	E log (1 + PsA) = − 11.44 + 0.483 log (1 + TC) + 0.046 NO_3 + 0.136 COND	0.93	132.3	3, 32
12 Mile Red Cross Beach	F log (1 + PsA) = − 0.32 + 0.016 ATEMP	0.23	4.29	1, 14

PsA	= *P. aeruginosa*/100 mL	WTEMP	= Water Temperature (degrees Celsius)	DAY	= Day of year (January 1 is 1)
TC	= Total coliform/100 mL	ATEMP	= Air Temperature (degrees Celsius)	TP	= Total phosphorus (ug P/L)
FC	= Fecal coliform/100 mL	COND	= Conductivity (micromhos/cm)	pH	= log hydrogen ion concentration
TPREV	= Total people in previous hour	DTKN	= Dissolved total Kjeldahl nitrogen (ug N/L)	NO_3	= nitrate plus nitrite (ng N/L)
BPREV	= Bystanders in previous hour				

Water temperature also probably affected *Pseudomonas* via user preference.

4.5 *PSEUDOMONAS AERUGINOSA* IN EARS

In 1978, as part of the intensive study of Gull Lake beach, the swimming class members were asked to participate in a study of the occurrence of *P. aeruginosa* in their ears. After obtaining the appropriate permission, swabs were taken both from ears of the swimmers before they went in swimming and immediately after swimming. These swabs were then analyzed for *P. aeruginosa*.

The time trend of *Pseudomonas* found in swimmer's ears (Figure 22) followed closely that of *Pseudomonas* found in water (Figure 19). Low frequencies of isolation were observed in June and July with much higher frequencies in early August.

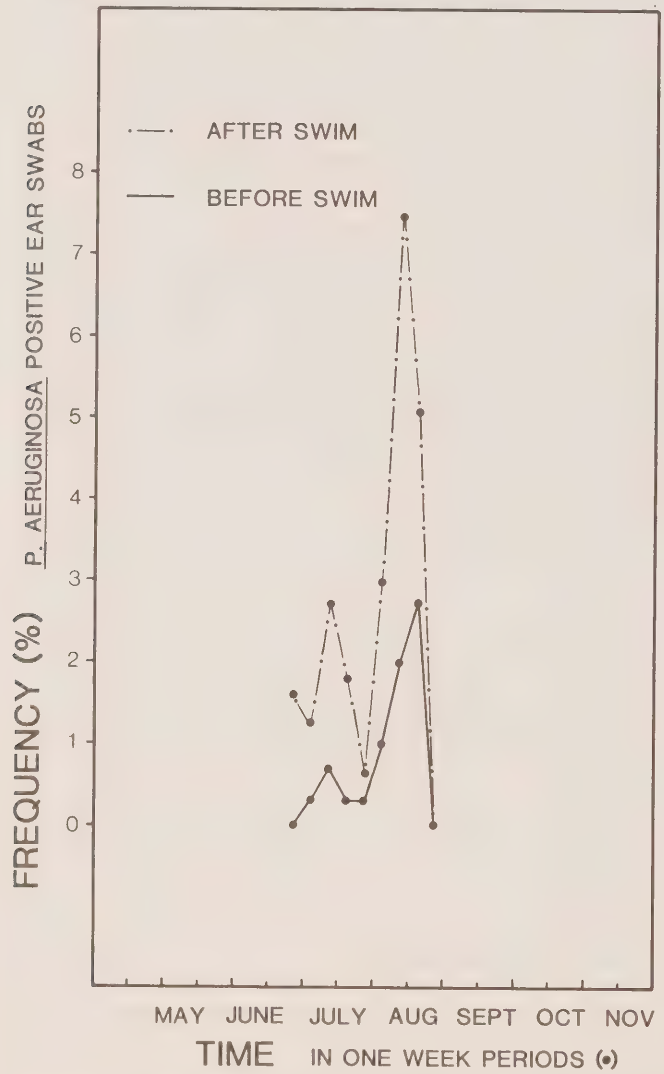


Figure 22: Time trend of frequency of isolation of *P. aeruginosa* from ear swabs of swimmers before and after swimming at Gull Lake beach during 1978.

Since the same swimmers' ears were swabbed both before and after swimming and over a number of days, the occurrence of *P. aeruginosa* in the ear could be followed over time. Some *Pseudomonas* were isolated prior to swimming from ears without any apparent signs of disease. Certain individuals had *Pseudomonas* present in an ear over an extended period of time. This apparent carrier state of an otherwise healthy individual is also confirmed in the literature (Seyfried and Fraser 1978; Young 1977). The original source of the *Pseudomonas* resulting in the carrier state cannot be determined. But the carrier state would form a source for *Pseudomonas* contamination of the beach waters.

More *Pseudomonas* were present in the ears after swimming than before. The duration of the swimming activity would be one and possibly two hours. Therefore, growth of the organism in the ear can probably be ruled out during this short period. The only source of the new *Pseudomonas* in the ears was water contaminated with *Pseudomonas* which gained access to the ear during swimming. Having gained access to the ear, the *P. aeruginosa* could go to the next step and cause ear infection

These data show that if the water in a swimming area contains *P. aeruginosa*, swimming activity will introduce the *Pseudomonas* into the ear of the swimmer. Whether that *Pseudomonas* will cause an ear infection will depend on a number of medical circumstances which are beyond the scope of this study.

Table 21. Frequency of *P. aeruginosa* serotypes in lake water, sediment, swimmers' ears and *otitis externa* patients' ears.

Serotype	Frequency for Source of Isolates (percent of source total)					
	Water	Sediment	Swimmers Ears	Patients Ears	Total	(Total No. Serotypes)
Untypable	13.4	14.6	6.2	15.4	13.3	(214)
0:1	10.8	9.2	10.3	13.5	10.5	(170)
0:11	10.1	6.7	3.1	3.8	8.8	(142)
0:4	7.6	7.6	3.1	5.8	7.3	(117)
0:6	5.1	7.6	6.2	5.8	5.7	(92)
0:5d	6.1	3.2	1.0	1.9	5.1	(82)
0:3, 4, 9, 10	5.9	1.3	0.0	0.0	4.5	(72)
0:10	4.4	1.3	6.2	5.8	3.9	(63)
0:6, 10	1.4	12.1	1.0	5.8	3.6	(58)
0:3, 4, 6, 9, 10	4.3	1.3	0.0	0.0	3.3	(53)
Self Agglutination	2.8	0.3	5.2	11.5	2.7	(44)
0:7	2.1	5.4	1.0	0.0	2.6	(42)
0:6, 10, 5d	1.7	4.5	3.1	1.9	2.3	(37)
0:3	1.6	1.9	6.2	7.7	2.1	(34)
0:4, 6, 9, 10	1.5	0.3	0.0	0.0	1.1	(18)
0:3, 4, 6, 10, 5d	1.0	0.0	3.1	3.8	1.0	(16)
0:9	1.1	1.0	0.0	0.0	1.0	(16)
0:4, 9, 10	1.4	0.0	0.0	0.0	1.0	(16)
0:10, 5d	0.5	2.5	1.0	0.0	0.9	(15)
0:1, 4	0.4	2.5	0.0	0.0	0.8	(13)
0:3, 4, 10, 5d	0.3	0.0	6.2	3.8	0.7	(12)
0:9, 10	1.0	0.0	0.0	0.0	0.7	(12)
0:4, 6	0.6	1.0	1.0	0.0	0.7	(11)
0:5c	0.0	3.2	0.0	0.0	0.6	(10)
0:4, 7, 9	0.9	0.0	0.0	0.0	0.6	(10)
0:7, 9, 10	0.4	1.6	0.0	0.0	0.6	(10)
0:4, 5d	0.5	0.0	3.1	0.0	0.6	(9)
0:7, 9	0.7	0.3	0.0	0.0	0.6	(9)
0:4, 7, 9, 10	0.7	0.0	0.0	0.0	0.5	(8)
0:3, 4, 7, 9, 10	0.6	0.3	0.0	0.0	0.5	(8)
Source Total	1149	314	97	52		1612

In addition to the serotypes listed in the table, the following serotypes are some which were recovered at less than 0.5 percent frequency:
(0:4, 9) (0:4, 7) (0:3, 4) (0:6, 5d) (0:4, 10) (0:12) (0:3, 6) (0:7, 10) (Psl1) (0:3, 10) (0:4, 5c) (0:6, 5c) (0:10, 5c) (0:1, 7) (0:6, 12) (S13) (0:5c, Psl1) (0:3, 5d) (0:6, 11) (0:1, 13) (0:1, 11) (0:3, 9) (0:1, 5c) (0:7, 5d) (0:8, 5d)(0:7, 8) (0:2a) (0:12, 5d) (0:11, 12) (S13, 5d).

4.6 IDENTIFICATION AND TYPING RESULTS FOR *PSEUDOMONAS AERUGINOSA*

Of the 1612 *P. aeruginosa* isolates serotyped (Table 21), 65.4 percent(1055) originated from Gull Lake beach. By source, 71.5 percent (822) of the water isolates, 45.5 percent (143) of the sediment isolates and 92.8 percent (90) of the ear isolates from swimmers were from Gull Lake beach.

At Gull Lake beach, serotypes 0:11 and 0:1 were the most common serotypes among the water isolates (Table 22). Serotypes 0:1 and 0:6 were most common serotypes among the sediment isolates. Serotype 0:1 was the most common serotype among the isolates from swimmers' ears. A large percentage of isolates from all sources could not be categorized because they either were untypable i.e., they did not react in the test system, or were self-agglutinating i.e., they gave a positive reaction to all tests in the system.

Table 22. Frequency of *P. aerugenosa* serotypes in beach water, sediment, and swimmers' ears at Gull Lake Beach.

Serotype	Frequency for Source of Isolates (percent of source total)					
	Water	Sediment	Swimmers Ears	Gull Lake Beach Total	Total	(Total No. Serotypes)
Untypable	10.8	15.4	6.7	11.1	13.3	(214)
0:1	13.7	10.5	10.0	13.0	10.5	(170)
0:11	13.9	5.6	3.3	11.9	8.8	(142)
0:4	3.0	7.7	3.3	3.7	7.3	(117)
0:6	6.6	9.8	6.7	7.0	5.7	(92)
0:5d	7.9	5.6	1.1	7.0	5.1	(82)
0:3, 4, 9, 10	5.4	1.4	0.0	4.4	5.1	(72)
0:10	5.5	2.1	4.4	4.9	3.9	(63)
0:6, 10	1.2	6.3	1.1	1.9	3.6	(58)
0:3, 4, 6, 9, 10	4.7	1.4	0.0	3.9	3.3	(53)
Self Agglutination	2.8	0.7	5.6	2.7	2.7	(44)
0:7	1.3	0.7	1.1	1.2	2.6	(42)
0:6, 10, 5d	2.1	1.4	3.3	2.1	2.3	(37)
0:3	1.3	0.7	6.7	1.7	2.1	(34)
0:4, 6, 9, 10	1.6	0.7	0.0	1.3	1.1	(18)
0:3, 4, 6, 10, 5d	1.3	0.0	3.3	1.3	1.0	(16)
0:9	0.4	1.4	0.0	0.5	1.0	(16)
0:4, 9, 10	1.2	0.0	0.0	0.9	1.0	(16)
0:10, 5d	0.5	4.2	1.1	1.0	0.9	(15)
0:1, 4	0.5	4.2	0.0	0.9	0.8	(13)
0:3, 4, 10, 5d	0.5	0.0	6.7	0.9	0.7	(12)
0:9, 10	0.7	0.0	0.0	0.6	0.7	(12)
0:4, 6	0.7	0.7	1.1	0.8	0.7	(11)
0:5c	0.0	6.3	0.0	0.9	0.6	(10)
0:4, 7, 9	0.0	0.0	0.0	0.0	0.6	(10)
0:7, 9, 10	0.0	0.0	0.0	0.0	0.6	(10)
0:4, 5d	0.5	0.0	3.3	0.7	0.6	(9)
0:7, 9	0.4	0.0	0.0	0.3	0.6	(9)
0:4, 7, 9, 10	0.0	0.0	0.0	0.0	0.5	(8)
Source Total	822	143	90	1055		1612

In addition to the serotypes listed in the table, the following serotypes are some which were recovered at less than 0.5 percent frequency: (0:4, 9) (0:4, 7) (0:3, 4) (0:6, 5d) (0:4, 10) (0:12) (0:3, 6) (Ps11) (0:3, 10) (0:4, 5c) (0:6, 5c) (0:10, 5c) (0:1, 7) (S13) (0:3, 5d) (0:6, 11) (0:1, 3) (0:1, 11) (0:3, 9) (0:11, 5c) (0:8, 5d) (0:7, 8) (0:5d, S13).

The most frequently encountered serotype at both shoreline and mid-lake locations was 0:4 (Table 23). At the beach locations, serotype 0:1 was most frequently isolated. Cultures which were untypable with the serotyping scheme accounted for 13.3 percent of all cultures typed. Untypable cultures were more frequent among isolates recovered from the shoreline and mid-lake locations than the beach locations. There was a higher percentage of the beach isolates which produced single serotypes. Of the *P. aeruginosa* recovered from beaches 52.2 percent produced single serotypes. Approximately 42 percent of cultures isolated from shoreline locations produced serotypes, as did 21.2 percent of cultures isolated from mid-lake locations.

Table 23. Frequency of common serotypes at various locations.

Serotype	Frequency at location (percent of location total)			
	Shoreline	Mid-lake	Beach	Total
Untypable	20.4	19.9	11.7	185
0:1	4.5	0.0	13.4	136
0:11	2.7	0.7	12.6	125
0:4	23.5	10.6	3.5	101
0:5d	0.9	2.6	7.8	79
0:3, 4, 9, 10	7.1	6.6	4.9	72
0:6	0.9	0.0	7.0	68
0:3, 4, 6, 9, 10	1.8	5.3	4.4	53
0:10	1.8	1.3	4.8	51
Self Agglutination	1.4	3.3	4.8	33
0:6, 10	4.0	0.0	1.8	26
0:7	5.0	0.7	1.3	24
0:3, 6, 10,	0.9	0.0	1.9	20
0:4, 6, 9, 10	0.5	2.0	1.5	18
0:9	1.4	5.3	0.5	16
0:4, 9, 10	0.9	2.6	1.1	16
0:3	1.4	0.0	1.3	15
0:3, 4, 7, 9, 10	0.9	5.3	0.4	14
0:1, 4	1.4	0.0	1.1	13
0:10, 5d	0.5	1.3	1.0	12
0:9, 10	0.9	2.6	0.6	12
0:3, 4, 6, 10, 5d	0.0	0.0	1.2	11
0:4, 6	1.4	0.0	0.7	10
0:4, 7, 9	0.9	5.3	0.0	10
Location Total	244	151	938	1315

Generally, the common serotypes were encountered, with varying frequencies from all three location types, shoreline, mid-lake, and beach. The differences between the frequencies of recovery of individual serotypes at each of the three locations may be attributable to the differences in the total number of cultures serotyped from each of the three locations.

4.6.1 DISCUSSION OF TYPING RESULTS

Serotypes 0:1, 0:11, 0:4, 0:6, 0:5d, 0:10, 0:3, and 0:3, 4, 6, 10, 5d (Table 21) were encountered with similar frequency in the ears of swimmers and of *otitis externa* patients (suggesting that the organisms from both sources were closely related). The frequency with which serotype 0:6, 10, serotype 0:4, 5d, self-agglutinating, and untypable isolates were encountered in the ears of swimmers differed from the frequency in the ears of *otitis externa* patients.

The role of water in the transmission of *P. aeruginosa* related diseases to humans, domestic animals, and plants is well documented in the literature (Hoadley 1977). However, confusion abounds over the normal habitat of this organism. One group of scientists favours plants and soil as the natural and permanent reservoir of the organism, whereas the other

group believes the human intestine and the resulting human sewage to be the primary source of the organism (Young 1977). Laboratory experiments have shown that *P. aeruginosa* may be able to grow in the environment in organically enriched surface waters where the water temperatures exceed 30°C. *P. aeruginosa* has seldom been recovered from cold surface waters except following rains or recent contamination by human feces or sewage (Hoadley 1977).

The frequency of recovery for the common serotypes, varied markedly from the beach location to the lake shoreline and mid-lake locations (Table 23). A higher percentage of the beach isolates belonged to serotypes 0:1, 0:11, 0:5d, 0:6, and 0:10 than either the shoreline or mid-lake isolates. Similarly, a higher percentage of the shoreline and mid-lake isolates belonged to serotype 0:4 than the beach isolates. The percentage of untypable cultures from shoreline and mid-lake locations were similar, whereas the percentage of untypable cultures from the beach was much lower. Of the isolates serotyped from the beach locations, 52.2 percent produced single serotypes, whereas 42.1 percent of the shoreline isolates produced single serotypes, and only 21.2 percent of mid-lake isolates produced single serotypes.

These differences in the frequency of individual serotypes and of total single serotypes were significant in that the procedures available to characterize *P. aeruginosa* serologically were developed for medical strains. If the *P. aeruginosa* originated from the swimmers themselves, than the differences in recovery frequency were logical. The *P. aeruginosa* recovered from the beach, having originated from the swimmers, were more closely related to medical isolates than are the *P. aeruginosa* recovered from lake shoreline or mid-lake locations where human recreational activities are less intense. It is not known whether the isolates recovered from lake shoreline and mid-lake locations were of human origin and with time underwent serological changes or whether they were innately different having originated from plants and soil.

The serotyping program showed that the *P. aeruginosa* recovered from both the ears of swimmers and of *otitis externa* patients to be closely related. The *P. aeruginosa* from beach waters and the ears of swimmers were also closely related. These results, therefore, suggested that the *P. aeruginosa* commonly encountered in bathing waters were likely of human origin and that water is a vector by which *P. aeruginosa* can be transferred from one swimmer to another.

Table 24. Comparison of serotyping results between the present study and those of Seyfried (1976) and Fraser (1977).

Serotype	Present Study %	Seyfried (1976) %	Fraser (1977) %
0:6	5.7	40.16	30.82
0:10	3.9	18.03	17.40
0:3	2.1	11.48	11.53
0:1	10.5	6.56	7.65
0:11	8.8	4.10	14.02
Untypable	13.3	1.64	2.78
Multiple:			
Beach	33.4	)	)
Shoreline	36.1	)0	)0.49
Mid-Lake	55.6	)	)
Total Number of isolates	1612	122	1006

The results of the serotyping programs can be compared with the works of Seyfried (1976) and Fraser (1977) who employed the same serotyping scheme (Duncan et al 1976) (Table 24). Of the 122 *P. aeruginosa* isolated from Muskoka lakes (Seyfried 1976) 40.16 percent were serotype 0:6, 18.03 percent were 0:10 and 11.48 percent were 0:3. Only 1.64 percent of these isolates were untypable. In the present study, the frequencies of serotypes 0:6, 0:10, and 0:3 were much lower and the frequencies of the untypable isolates were much higher. Seyfried encountered no multiple serotypes, whereas in this study, 36.1 percent of shoreline cultures and 55.6 percent of mid-lake cultures displayed multiple serotypes. Fraser (1977) examined *P. aeruginosa* in Lake Ontario, sewage effluent, swimming pools, and ears. Serotypes 0:6, 0:1, 0:11, 0:3, and 0:10 were recovered from the environmental sources with the highest frequencies. From the ear isolates, serotypes 0:6, 0:3, 0:1, 0:10 and 0:11 were recovered most often. In the present study of *P. aeruginosa* isolates recovered from outer ears of swimmers, serotypes 0:1, 0:6, 0:10 and 0:3 were encountered most frequently.

Cultures which yielded multiple serotypes accounted for 33.4 percent of beach isolates, 36.1 percent of shoreline isolates, and 55.6 percent of mid-lake isolates in this study while no cultures yielding multiple serotypes were encountered by Seyfried and 0.49 percent of cultures serotyped by Fraser yielded multiple serotypes. Generally where isolates give positive reactions to two or more serotypes, the results are not as reliable for tracing purposes as those from cases where isolates react positively for only one serotype (Duncan et al 1976). A variety of factors can cause the occurrence of multiple serotypes (Homma et al 1972; Pitt and Erdman 1977).

Although Seyfried and Fraser employed the same serotyping scheme in their respective studies of *P. aeruginosa*, the testing anti era were slightly different. Duncan et al (1976), however, conducted a study in which the two sets of antisera were shown to be comparable in performance.

The present study demonstrated the wide distribution of many *P. aeruginosa* serotypes. No serotype of *P. aeruginosa* was observed to be unique to any one geographical location or to an environmental or human source. The results of the serotyping program combined with the results of the ear swabbing program and the water quality assessment programs support the thesis that the major source of *P. aeruginosa* in bathing waters is the swimmers themselves with the water as the vector for transmission from one swimmer to another.

4.7 INFECTION CASE RATE VERSUS *P. AERUGINOSA* LEVELS

On all questionnaires, there was at least one question which required the respondent to give a list of locations at which they swam and three to five spaces were provided for a write-in answer. This question was then coded so that all questionnaires could be categorized by the locations at which the respondent swam. The ear infection case rate for those persons who swam at a particular location can be compared with that for those who did not report swimming at that location and with the general case rate across all questionnaires (Table 25).

Table 25. Case rates of *otitis externa* ear infection in users of the 1979 study lakes and beaches.

Lake/Beach (1979)	Case Rate/100 Users: Total Body of Water (Lake and Beach)	Beach only	Lake only
Gull	10.4	11.7	8.6
Head	4.3	3.6	10.0
Pine	11.3	11.9	8.0
Mary	8.0	8.3	8.1
Vernon	7.3	8.0	0.0
12 Mile	9.5	-	8.7
1) Public	-	0.0	-
2) Red Cross	-	10.8	-
Muskoka	6.3	-	-
General Response	7.6		

number of questionnaires summarized = 995.

In 1979, Gull Lake beach, Pine Lake beach and Red Cross beach on 12 Mile Lake gave higher case rates than were seen at the corresponding lakes or the general response. Head Lake beach and Public beach on 12 Mile Lake gave aberrantly low case rates. At all other locations reported on the 1979 questionnaires there were too few responses to give an accurate picture of the case rate.

From the 1978 questionnaire, two bodies of water, Gull Lake and Muskoka Lake, reported a sufficient number of questionnaires to give a case rate. They gave case rates of 14.7 and 9.0 respectively per 100 users as compared to a general response case rate of 10.0 per 100 users. The Gull Lake beach data were not separated from the rest of the Gull Lake data.

From the 1978 and 1979 case rates, Gull Lake beach, Pine Lake beach and 12 Mile Lake Red Cross beach contributed to an increased level of ear infection among the user population. Because the summer was used as the base for time, the risk of contracting an ear infection by using the swimming location was assumed uniform over the entire period. However, the risk was seasonally variable as seen from the dates when users acquired an ear infection. Therefore, the summer case rate will be underestimating the maximum case rates that could be expected at mid-summer and peak-use periods.

The *P. aeruginosa* concentrations which can be associated with a particular risk of contracting the disease or a particular case rate need to be examined from two perspectives: the concentrations actually associated with a given level of disease incidence and the underlying mechanism of acquiring the disease. In the case of *P. aeruginosa*, the disease causing organism is being measured directly, instead of through a secondary indicator such as fecal coliform. Therefore, if the disease causing organism is present, a risk of contracting the disease exists and the concentration of the organism is a measure of the magnitude of that risk. This means that the risk for a particular individual is related directly to the concentration of the organism to which he is exposed and not to some measure of an average state. The extremes of the distribution of concentration and especially the high values are, therefore, important in the assessment of the risk to an individual.

In 1979, the median concentration of *P. aeruginosa* at all locations was 0 bacteria per 100 mL since less than 50% of the samples contained any *P. aeruginosa* (Table 7). Locations could not be distinguished from each other by their average state, even if there was a difference in the disease case rate. The maximum concentration of *Pseudomonas* fluctuated radically with both location and time. The 75 percentile concentration of *P. aeruginosa* provided a realistic and more stable measure of the high concentration found in any time period. The 75 percentile concentration is defined as that concentration below which 75% or three-quarters of the concentrations are observed. The 75 percentile *Pseudomonas* concentrations for most shoreline locations was 2 bacteria per 100 mL or less (Figure 14). However, for beach locations, the 75 percentile concentration regularly exceeded 10 *Pseudomonas* per 100 mL and occasionally went into the range 100 to 1000 *Pseudomonas* per 100 mL (Figure 17). The 75 percentile *P. aeruginosa* concentrations did reflect the differences in disease case rate seen between lake shoreline and beach locations and therefore, the risk of contracting an ear infection.

5. CONCLUSIONS AND IMPACTS

In the introduction, a goal of the Microbiology Component of the Lakeshore Capacity Study was to define and quantify the influences of lakeshore development and the associated recreational activities upon the bacteriological quality of the lake water and the infections among the human users of the water. Other microorganisms are important to the public health of the people in the study area. The importance of maintaining adequate treatment of the water used for drinking, washing and food preparation and of the sewage produced by a development cannot be stressed strongly enough. In the examination of any proposal for development, these areas must be considered if the health of the people is to be safeguarded.

In this study, the research was restricted to one organism, *P. aeruginosa*, and one disease, an ear infection called *otitis externa*. As a result of our study, a number of component parts of a model presented as a flow diagram in Figure 1 have been examined. This model is supported by the general microbiological literature, by the general inclination of microbial ecology and by the findings of our study.

In the Muskoka-Haliburton lakes *P. aeruginosa* was virtually absent except where human activity was great, e.g. at a beach. The regression analyses and the time trend of isolation of *Pseudomonas* suggested that temperature played a major part in the occurrence of *P. aeruginosa*. *Pseudomonas* was more frequently recovered as temperatures in the lake water increased above 20°C into the range 24 to 25°C, the period of time between mid-July and mid-August. The question of nutrients and organic enrichment and their effects on *Pseudomonas* levels was not resolved by this study.

In the current study, up to 3% of the swabs from healthy swimmers' ears taken before swimming and up to 8% of the swabs from healthy swimmers' ears after swimming activity were positive for *P. aeruginosa*. When the swabs submitted by local physicians from diagnosed cases of *otitis externa* were examined, 58% showed presence of *P. aeruginosa*. The swimmer and cottage questionnaires showed that between 8% and 12% of the swimming population experienced an ear infection during the summer months. The control population, which did not swim, were not represented because the survey was carried out mainly at the beach. In the ear swabbing portion of the study, 33 members of the swimming class at Gull Lake beach at one point in time had *Pseudomonas* present in their ears. Of the 22 subsequently completing the questionnaire, 3 (13%) reported that they developed an ear infection. Both the incidence of ear infections and the levels of *Pseudomonas* in the water were highly seasonal. The levels of *Pseudomonas* were very low (0 to 30 per 100 mL) and very highly variable (a few samples with up to 1000 per 100 mL).

In summary, the combination of results reported in the literature and the findings of this study can place values on the levels of occurrence of *P. aeruginosa* in each of the main boxes of the model. About 10% of the normal healthy human population carry *Pseudomonas* in their gastro-intestinal system and excrete it in their feces. In 'northern temperate surface waters' not affected by humans, *P. aeruginosa* levels are extremely low and virtually absent. These levels are affected by water temperature and the amount of human influence. *P. aeruginosa* is found in a low percentage of normal healthy ears, 10 to 14% (Hoadley and Knight 1975), 1% (Singer et al 1952) and 3% (this study). This study suggested that this percentage was increased when measured after swimming. The *otitis externa* ear infection was between 7 and 12% for people who swam. Of the *otitis externa* cases, 58% showed *P. aeruginosa*.

However, this study did attempt to go beyond the simple enumeration of *P. aeruginosa* and to examine some of the interactions.

In the 1978 Gull Lake beach study, the following multiple linear regression equation was developed:

$$\log(\mathrm{PsA} + 1) = -0.72 + 0.041\,\mathrm{WTEMP} - 0.164 \log(\mathrm{FC} + 1) + 0.0048\,\mathrm{TPREV}$$

where FC and PsA were the levels of fecal coliform and *P. aeruginosa* per 100 mL, WTEMP was the water temperature in degrees Celsius and TPREV was the total number of users in the previous hour at the swimming area. This relationship was not obtained when the study was repeated at the same location the following year or at different locations. This non-reproducibility was probably due to the extreme variability of the data, erroneous statistical assumptions and the very low level of the *Pseudomonas* in the surface water.

At Gull Lake beach in 1979, the people who swam at the beach developed ear infections at a rate of 12% or 12 infections per 100 users. At this beach, 75 percentile levels of 10 *Pseudomonas* per 100 mL and greater were present on a number of occasions. In the case of cottaged shoreline areas, the people developed ear infections at a rate of 8% or less. The 75 percentile levels of 2 or less *Pseudomonas* per 100 mL were usual. The important levels of *P. aeruginosa* where the effect of increased ear infections can start to be seen are in the range of 2 to 10 *Pseudomonas* per 100 mL when examined as a 75 percentile value.

If a water quality criterion value was wanted, a 75 percentile *P. aeruginosa* concentration of 10 bacteria per 100 mL would be the best current choice, since above that value a noticeable increase in the disease rate was present. Operationally, this

water quality criterion value could be used in the following fashion: If more than one quarter of the samples taken at a particular location contain more than 10 *P. aeruginosa* per 100 mL, then the water quality criterion has been exceeded and there is a significant risk to swimmers of getting an ear infection. The action to be taken at this occurrence would be to limit the number of swimmers since the swimmers are the source of the bacteria. However, since the results of the sample analyses are not available immediately, this action could only be applied when the location in question historically has the elevated levels of *Pseudomonas*. From this study, this action would rarely be required for most lakeshore cottage development but may be required for heavily used bathing beaches.

5.2 IMPACT OF THE MODEL AND THE STUDY RESULTS

The results of the Lakeshore Capacity Microbiology Component study and the literature suggest that the model which was proposed can be validated. However, the current results do not allow the model to be stated as a fully quantified mathematical model, but only as a qualitative model with some sections having quantitative data. The results and the model do provide a basis for suggesting a *P. aeruginosa* water quality criteria for use in Ontario and design criteria for swimming areas in order to minimize the risks of *otitis externa* ear infections.

When the 75 percentile concentration of *P. aeruginosa* exceeded 10 bacteria per 100 mL, there was a noticeable increase in the incidence of ear infections among the swimmers at that location. This level of bacteria contamination can, therefore, be set as a water quality criterion. When the *Pseudomonas* concentrations in a swimming area regularly exceed this water quality criterion, the users of the area have a greater risk of contracting an ear infection and the use of the swimming area should be restricted until the cause of the situation has been clarified and/or rectified.

When designing a swimming area in association with a cottage development, the design should also take into consideration the risk of bacterial contamination. The swimming area should be located away from and upstream of possible sources of sewage and sewage impacted soil runoff. It should be located away from shallow, organically rich, marshy areas which may be warmed by the sun or by a thermal input and should be designed to have a good flushing or dilution with water from outside the impacted area. Human users should be dispersed over as wide an area as possible. These factors in the design are intended to reduce the intensity of swimmer use or to reduce the *P. aeruginosa* levels and thereby reduce the likelihood of *otitis externa* ear infections.

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7. APPENDIX A

This appendix contains a copy of the various questionnaires, forms and descriptive letters used by the Microbiology Component in the years 1978 and 1979.

1978

1) Letter and questionnaire for swimmers and beach users

1979

1) Letter, instruction sheet and card questionnaire for beach users
2) Physician questionnaire and letter
3) Patient questionnaire and letter

Ontario

Ministry of the Environment

135 St. Clair Avenue West
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Microbiology Laboratory #8
P.O.Box 58
Dorset, Ontario
P0A 1E0

May 5, 1978

Dear Sir or Madame,

Each year numerous swimmers encounter ear infections associated with the bacteria Pseudomonas aeruginosa. This organism has been associated with ear infections (otitis externa) of users of swimming pools and because it is often encountered in lake water, it may also be involved as a causative agent of ear infections of swimmers at recreational lakes. For these reasons the Ontario Ministry of the Environment is conducting an extensive study of bacterial ear infections associated with swimmers at recreational lakes in the Muskoka area.

In order to gather information concerning the swimming habits and incidence of disease of users of recreational waters, we request your cooperation in providing us with this information. In the month of August, with your permission, we will send to you a questionnaire concerning your swimming habits and experience with ear infection during the summer months. Additional copies of this questionnaire will be provided at that time for members of your family who also may wish to participate in the study. Along with these questionnaires you will receive stamped, self-addressed envelopes to provide a convenient means for you and your family members to return the completed questionnaires. Answers to these questions from both swimmers and non-swimmers are essential to the success of the study and will greatly assist the Ministry of the Environment in its efforts to determine the corrective measures required to eliminate potential health hazards in lake waters used for recreational purposes. Information gathered from participants in this study will be kept strictly confidential.

A sample questionnaire has been provided with this letter to convey examples of the type of information that we will request from you near the end of the summer. We hope that this sample questionnaire will assist you in maintaining a record of your general swimming habits during the summer months.

We believe that this research will demonstrate the significance of Pseudomonas aeruginosa as an indicator of health risks to users of recreational lakes and that the findings will be of interest to all swimmers. For your participation in this important study we are sincerely grateful.

Sincerely yours,

Allan Burger

A. Burger, Supervisor,
Microbiology Section.

Karl Lautenschlager

K. Lautenschlager,
Scientist,
Lakeshore Capacity Study.

MINISTRY OF THE ENVIRONMENT
Microbiology Laboratory #8
P.O. Box #8
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SWIMMER'S AND NON-SWIMMER'S QUESTIONNAIRE

Please answer all questions that pertain to your personal situation by supplying the appropriate information or by circling the most appropriate typed answer. After completing the questionnaire, please enclose it in the attached envelope and forward it by mail.

1. What is your Name? ______________________________

2. What is your sex? Male ______ or ______ Female.

3. What is your age? ______ years.

4. Are you a swimmer or a non-swimmer? ______ Swimmer or ______ Non-Swimmer.

5. If you are a non-swimmer, do you usually wade or otherwise splash about in waters used by swimmers? ______ Yes or ______ No.

6. At what bodies of water did you swim or wade during this summer? Please indicate the names of the water bodies and also the type of swimming facilities at each site (e.g. private pool, public beach, river, lake shoreline, cottage, etc.).

Name of Water Body	Type of Swimming Facilities	
A ______________	______________________	A
B ______________	______________________	B
C ______________	______________________	C
D ______________	______________________	D
E ______________	______________________	E

7. At which body or bodies of water did you swim or wade most frequently? ______________

____

8. How often did you usually swim during the summer months? ______ Times per day; ______ Times per week.

9. When you did swim or wade for how long did you usually remain in the water? ______ hour(s) per swim; ______ hour(s) per day.

10. On the average, how many hours per week did you spend swimming during the month of May, June, July and August? ......

    In May ______ ; In June ______; In July ______; In August ______.

11. During each of these months, on days when you swam, at what time(s) of day or night did you usually swim? ______ (Please indicate AM or PM)

    In May ____________________; In June ____________________;

    In July ____________________; In August ____________________.

12. On what day(s) of the week during each month did you usually swim?

Month	Monday	Tuesday	Wednesday	Thursday	Friday	Saturday	Sunday
	Days of the Week						
May	1	2	3	4	5	6	7
June	1	2	3	4	5	6	7
July	1	2	3	4	5	6	7
August	1	2	3	4	5	6	7

13. At the sites at which you most often swam this summer, approximately how many other swimmers were usually present in the water at the times when you were swimming?

Swimming Sites	None;	a Few;	Several;	Many	Very Many
	Approximate Number of Other Swimmers Present				
______________	1	2	3	4	5
______________	1	2	3	4	5
______________	1	2	3	4	5

14. Do you consider the sites at which you swam to be of good water quality? _______ Yes or _________ No.

	Yes	No
15. Do you dive?	Y	N
16. Do you swim under water?	Y	N
17. Do you wear a bathing cap?	Y	N
18. Do you wet your head while swimming?	Y	N
19. Do you wear ear plugs or other forms of ear protection while swimming?	Y	N
20. Have you ever experienced an ear infection?	Y	N
21. Do you often experience ear infection?	Y	N
22. Did you experience an ear infection this summer?	Y	N

** The following questions should be answered only by swimmers who have experienced ear infection.

	Yes	No
23. If you experienced an ear infection this summer did you associate the infection with your swimming activities?	Y	N
24. If you experienced an ear infection this summer did you seek medical treatment?	Y	N
25. If you experienced previous ear infections were they treated by a physician?	Y	N
26. Have you ever been hospitalized for an ear infection?	Y	N
27. If you experienced an ear infection this summer were you already receiving medication for another ailment when the symptoms of ear infection appeared?	Y	N

28. If you experienced previous ear infections on what date(s) or at what time(s) of year did they occur? ....

____

29. If you experienced an ear infection this summer on what date (approximately) did the symptoms first occur? ....___________________________

** The following questions should be answered only by swimmers who experienced an ear infection this summer.

30. In the two-week period just before the onset of the symptoms of ear infection at what water bodies had you been swimming? ...... ___________________________

Site 1: ________________ Site 2: ________________ Site 3: ________________

31. During the two-week period just prior to the onset of symptoms how many times had you been swimming and for how long did you stay in the water usually? ...... ____________ Times swimming; ______________ hour(s) in the water on each swimming occasion (average).

32. For how long did you experience ear infection symptoms before you obtained medical treatment: ..... _________ days; _________ weeks.

33. In what community did you obtain medical treatment? ..... ___________________________

34. Any comments concerning the waters in which you swam: ___________________________

____

____

We express our sincere gratidude for your cooperation in this study.

Ontario

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of the
Environment

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Microbiology Laboratory #8
P.O. Box 58
Dorset, Ontario   P0A 1E0

Tel. 705-766-2712

June 20, 1979

Dear Patient or Guardian;

The Ontario Ministry of the Environment is conducting a investigation into the incidence and causes of ear infections associated with the bacterium, Pseudomonas aeruginosa. The Ministry is particularly interested in obtaining information from individuals who have suffered from ear infections that may be related to the presence of this bacterium in recreationally-used lake water. To obtain the necessary information, we would very much appreciate your cooperation by completing the attached questionaire.

The Patient's Survey of Ear Infection (Otitis Externa) provides us with information concerning your swimming activities and the history of your ear infection. All answers on the questionaire are very important to the success of the study. Your doctor will also be providing us with some of detail of the symptoms and the prescribed treatment but we need you to provide some details about your current swimming habits. The information that you provide will be kept in the strictest confidence and only code numbers, never the names, will be used in reports concerning the findings of the investigation.

A stamped, self-addressed envelope has been provided with the questionaire for your convenience. Your cooperation in our study is sincerely appreciated.

Sincerely yours,

Allan Burger

A. Burger, Supervisor
Microbiology Section

Karl Lautenschlager

K.P. Lautenschlager, Scientist
Lakeshore Capacity Study

Ontario

Ministry of the Environment

135 St. Clair Avenue West
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Microbiology Laboratory #8
P.O. Box 58
Dorset, Ontario POA 1EO

CODE:________/________

CONFIDENTIAL

Physician's Survey of Otitis Externa

1. Date of treatment .... ______/______/ 1979
   Day Month

2. Date of onset of symptoms .... ______/______/ 1979
   Day Month

3. Describe the symptoms at the time of diagnosis (circle the appropriate answer)

Pain:-	None	Slight	Moderate	Severe
Swelling:-	None	Slight	Moderate	Severe
Discharge:-	None	Slight	Moderate	Severe

4. Ear(s) infected:- Left Right Both

5. Does the patient have a history of ear pathology? Yes No

6. Is there a concomitant infection? Yes No

7. Was there any recent prior antibiotic treatment? Yes No

8. Was hospitalization required for this case? Yes No

9. Comments:

Ministry of the Environment
Ontario

NAME________________________________
Surname Given Names

CONFIDENTIAL

Survey__________ 1. Age (years)__________

Date__________, 1979 2. Sex: M F

Circle the appropriate answer to questions:

3. TOURIST LOCAL COTTAGER RESIDENT

4. Swim: NEVER OCCASIONALLY FREQUENTLY VERY

5. Head: Y N	9. Hosp: Y N	13. Assoc: Y N
6. Class: Y N	10. Self: Y N	14. Sites: Y N
7. Ever: Y N	11. This: Y N	15. Participate:
8. Treat: Y N	12. Treat: Y N	Y N

16. First lake swimming __________, 1979

17. Name sites: A__________ B__________

C__________ D__________ E__________

22. Date previous onset: __________

23. Date this summer onset:__________, 1979

24. Community treated: __________

25. Doctor: __________

SUMMER MAILING ADDRESS PERMANENT ADDRESS

__________ __________

__________ __________

TEL. NO. TEL. NO.

Ontario

P.O. Box 58
Dorset, Ontario
P0A 1E0

Ministry
of the
Environment

The Ontario Ministry of the Environment as part of its Lakeshore Capacity Study is investigating the relationship, if any exists, between water quality and an outer ear infection (otitis externa), commonly known as "swimmer's ear". As part of our investigation we are asking questions of individuals at a number of locations in the Muskoka-Haliburton area. Would you take a few minutes to answer our questions? All information obtained will be strictly confidential and will only be used for the purpose of this study.

(Answers to all questions are to be recorded on the answer card)

I Name -surname, given names

1. Age -in years

2. Sex -male (M) or female (F)

3. Which of the best describes you:

   TOURIST -you are visiting in the area

   LOCAL COTTAGER -you reside within 30 miles of this location for only part of the year

   RESIDENT -you reside within 30 miles of this location year round

4. Which of the following best describes the frequency of your swimming activities:

   NEVER -you are a non-swimmer and do not go swimming

   OCCASIONALLY -you swim up to once, maybe twice, a week, and usually for less than one hour

   FREQUENTLY -you regularly swim between 1 and 4 times per week

   VERY -you regularly swim 4 or more times per week

5. Does your head, especially the area around your ears, get wet while you are swimming?

6. Are you enrolled in a swimming class at this location?

7. Have you ever had an outer ear infection (otitis externa)?

8. Have you ever been treated by a doctor or an emergency ward for an outer ear infection (otitis externa)?

9. Have you ever been hospitalized for an outer ear infection (otitis externa)?

10. Have you ever treated yourself for otitis externa without consulting a medical person using medication from a previous infection or from a drugstore shelf?

11. Have you had an outer ear infection (otitis externa) this summer?

12. Were you treated by a doctor or an emergency ward for otitis externa this summer?

13. Did you associate your ear infection with your swimming activity?

14. Do you regularly swim at more than one beach over the summer months?

15. Would you agree to participate further in this study by answering an additional questionaire that would be mailed to your home later this summer?

16. When (date) was your first swim in lakewater this summer?

17. Where do you regularly swim during the summer? Name the locations including pools. (Note: five spaces have been provided for your answers but all five do not have to be used)

22. When you previously had otitis externa what was the approximate date of the infection's onset include the year?

23. This summer what was the date of the infection's onset?

24. In what community (eg. town, city) did you obtain medical treatment for otitis externa this summer?

25. What was the doctor's name?

II What is your summer mailing address and telephone number?

III What is your permanent address and telephone number?

Thank you for answering our questions. If you have any questions please feel free to contact us.

Ontario

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135 St. Clair Avenue West
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Microbiology Laboratory #8
P.O. Box 58
Dorset, Ontario P0A 1E0

June 13, 1979

Dear

Each year many swimmers encounter the ear infection, otitis externa, after bathing in recreational waters. Pseudomonas aeruginosa, the bacterium associated with most of these ear infections is frequently detected in lake water used by humans. The Ontario Ministry of the Environment has been studying the relationship which exits between the quality of lake waters and the ear infections of bathers using the Muskoka-Haliburton region as a test area.

In order to determine the incidence of swimming-related otitis externa in recreational areas and to gather information regarding individual cases of this infection, the assistance of physicians in Bracebridge, Gravenhurst, Haliburton, Huntsville and Minden is being requested.

In the lecture slide presentations given to many of the local physicians this spring, some of our preliminary epidemiology findings were reported. The relationships among bather activity, water quality, and the incidence of otitis externa are likely to be established with additional data collection and analysis. However, the success of the study in establishing these relationships will depend on the accuracy of the estimates of the incidence of otitis externa among swimmers in the study area. Some of the information required can only be obtained from the members of the medical profession in the area. We are therefore requesting your participation in the study by providing us with medical information about the otitis externa patients treated this summer and by encouraging each patient to complete a questionaire. Please note that this year we are NOT requesting that ear swabs be taken for our study.

For each otitis externa patient treated, complete the brief Physician's Survey of Otitis Externa form which will serve to confirm the ear infection as a case of otitis externa and will provide us with important information about the patient's symptoms and history of ear ailments. The Patient's Survey of Ear Infection form is to be given to each otitis externa patient and is designed to gather information concerning the patient's swimming activities and the history of this and previous ear infections. This form can be completed and returned by the patient in the self-addressed stamped envelope provided. Both questionaires bear a specific code to assist us in compiling the data as it is returned.

These questionaires will be supplied to your office and to the emergency ward treatment areas of the hospitals in Bracebridge and Huntsville so that you will have ready access to them. The completed physician's questionaires will be picked by our staff on a regular basis until the end of the study in late September.

Your cooperation in this research will be greatly appreciated. If you have any questions concerning this year's study or if you wish to obtain summaries of our earlier study findings please contact us in Toronto (1-416-248-3008) or in Dorset (766-2712). We look forward to working with you over the summer months.

Sincerely yours,

Allan Burger, Supervisor,
Microbiology Section

Karl Lautenschlager, Scientist,
Lakeshore Capacity Study

Ontario

Ministry
of the
Environment

135 St. Clair Avenue West
Suite 100
Toronto, Ontario
M4V 1P5

CONFIDENTIAL

Code: ________/________

Microbiology Laboratory #8
P.O. Box 58
Dorset, Ontario POA 1E0

PATIENT'S SURVEY OF EAR INFECTION (OTITIS EXTERNA)

1. Name: ____

2. Age: __________(years)

3. Sex: MALE FEMALE

4. Summer Mailing Address — Permanent Mailing Address

______________________ ______________________

______________________ ______________________

______________________ ______________________

5. Telephone Number — Telephone Number

______________________ ______________________

6. Date of treatment for otitis externa ________/________,1979

7. Date of the start of the symptoms ________/________,1979

8. Are you a SWIMMER NON-SWIMMER (circle answer)

9. In the 2 weeks prior to the onset of symptoms of otitis externa where did you swim?

Name of Body of Water	Description (eg. cottage, pool, beach)
A______________________	______________________
B______________________	______________________
C______________________	______________________

10. How many other bathers were present in the water when you were swimming at the above locations? (circle the most appropriate answer)

A..........	NONE	1-20	20-200	MORE THAN 200
B..........	NONE	1-20	20-200	MORE THAN 200
C..........	NONE	1-20	20-200	MORE THAN 200

11. On what days of the week do you usually swim? (circle appropriate answers)

MONDAY TUESDAY WEDNESDAY THURSDAY FRIDAY SATURDAY SUNDAY

12. Do you dive?.. YES NO

13. Do you swim under water?.................................... YES NO

14. Do you wear a bathing cap?.................................. YES NO

15. Does your head get wet while you are swimming?......... YES NO

16. Do you wear ear plugs or other ear protection?............ YES NO

17. Do you associate the ear infection with swimming?...... YES NO

18. Have you previously had an ear infection?.................... YES NO

19. Have you ever been hospitalized for an ear infection?.. YES NO

20. Have you ever treated yourself for otitis externa with medication prescribed for a previous ear infection?...... YES NO

21. Have you ever treated yourself for otitis externa without obtaining medical advice?............................ YES NO

22. Have you ever suffered the loss of your recreational enjoyment because of an ear infection?....................... YES NO

23. Do you use untreated lake or river water for washing or bathing at your home or cottage?........................... YES NO

24. If you had a previous ear infection when did it occur, - month, time of year?

____

25. Please use the remaining space for any comments:

____

____

____

____

____

____

____

Your participation in this research is gratefully acknowledged.

Ontario

Ministry
of the
Environment

135 St. Clair Avenue West
Suite 100
Toronto, Ontario
M4V 1P5

Microbiology Laboratory #8
P.O. Box 58
Dorset, Ontario P0A 1E0

Tel. 705-766-2712

June 20, 1979

Dear Sir or Madame,

The Ontario Ministry of the Environment is conducting an investigation into the causes and incidence of ear infections associated with the bacterium, Pseudomonas aeruginosa. Previous studies at swimming pools have shown this bacterium to be associated with ear infections known as otitis externa or "swimmer's ear". Because this bacterium may be found in lake waters the Ministry is particularly interested in obtaining from the local residents, cottagers and swimmers concerning their swimming activities and their experiences with ear infections during the summer months.

In the previous two years the microbiologists from the Dorset field laboratory have conducted surveys at several lakes and beaches in Muskoka-Haliburton. From these surveys much new information has been obtained and our understanding of the factors involved in the transmission of bacterial infections through lake water has been enhanced. This year several sites in Muskoka-Haliburton will be examined in a effort to bring the study to a successful conclusion and to firmly establish the relationship between recreational activity/development and the incidence of ear infections.

While we will getting much information from the analysis of water and sediment samples collected at each study site, the information which is essential to the study can only be obtained from the people who use the lake water for swimming and other recreational activites. For this reason we request that you participate in the study by completing the attached questionaire. For your convenience the completed questionaire may be returned directly to the technician from whom it was obtained or, if necessary, a postage-paid mailing envelope will be provided. All answers to this questionaire are very important to the success of this study. It is hoped that, with your participation in the study, practical solutions to this public health problem in recreational water can be formulated.

The information gathered in the questionaires and interviews will be kept in the strictest confidence and only code numbers, never names, will be used in reports from the study. If you you have any questions concerning the study please contact us. Your cooperation in this important research is sincerely appreciated.

Sincerely yours,

Allan Burger

A. Burger, Supervisor,
Microbiology Section

Karl Lautenschlager

K.P. Lautenschlager, Scientist,
Lakeshore Capacity Study

Ministry of
Municipal
Affairs

Ontario

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1986
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LAKESHORE CAPACITY STUDY

TROPHIC STATUS

Government
Publications

LAKESHORE CAPACITY STUDY

TROPHIC STATUS

MAY 1986

Prepared by:
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Ministry of the Environment

Ontario
Ministry of
Municipal
Affairs
Bernard C. Grandmaître, Minister

Research and
Special Projects
Branch

The following Lakeshore Capacity Study reports

Committee Report
Land Use
Fisheries
Microbiology
Trophic Status
Wildlife
Integration

are available from:

Ministry of Municipal Affairs
Research and Special Projects Branch
777 Bay Street
13th Floor
Toronto, Ontario M5G 2E5

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LAKESHORE CAPACITY STUDY

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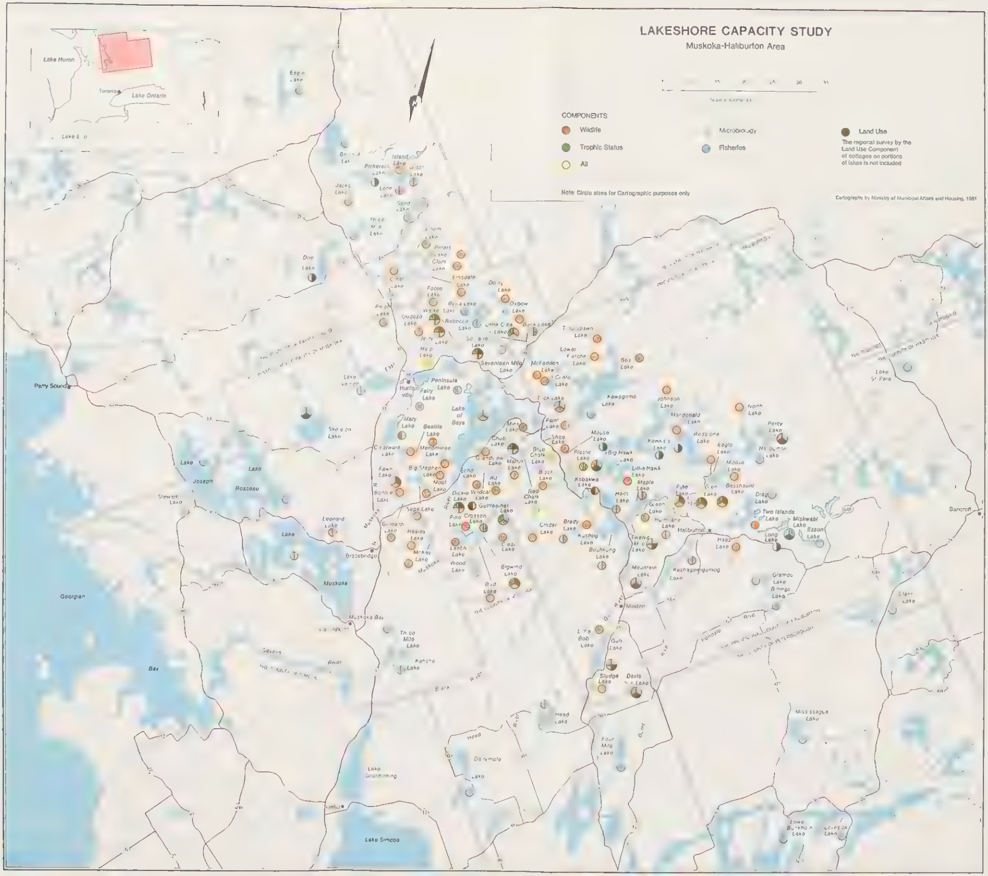
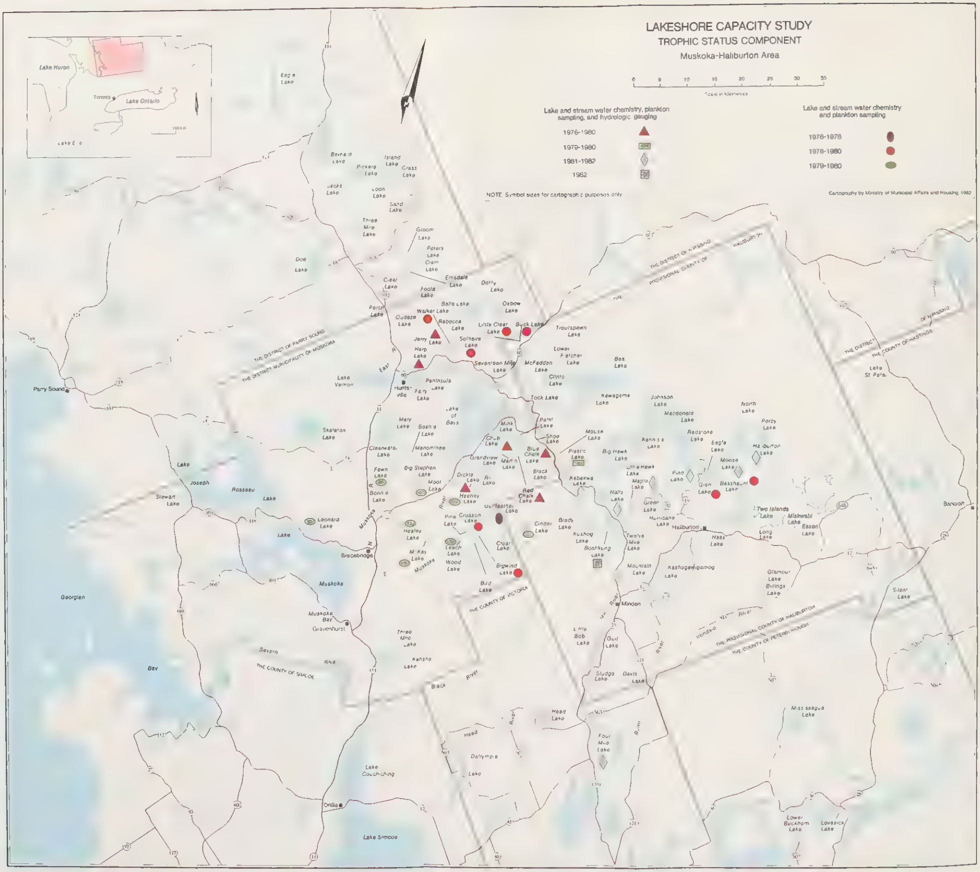

LAKESHORE CAPACITY STUDY
Muskoka-Haliburton Area
COMPONENTS
Wildlife
Trophic Status
All
Microbiology
Fisheries
Land Use
The regional survey by the Land Use Component of cottages on portions of lakes is not included
Note: Circle sizes for Cartographic purposes only
Cartography by Ministry of Municipal Affairs and Housing, 1981
Lake Huron
Toronto
Lake Ontario
Parry Sound
Huntsville
Bracebridge
Muskoka Bay
Haliburton
Minden
Bancroft
Georgian
Bay
Lake Joseph
Lake Rosseau
Lake Muskoka
Lake of Bays
Lake Simcoe
Island Lake
Peninsula Lake
Fairy Lake
Seventeen Mile Lake
McFadden Lake
Kawagama Lake
Johnson Lake
Percy Lake
Two Islands Lake
Miskwabi Lake
Esson Lake
Kushog Lake
Kashagawigamog Lake
Mountain Lake
Bigwind Lake
Crosson Lake
Gullfeather Lake
Sudge Lake
Davis Lake
Head Lake
Three Mile Lake
Kahshe Lake
Leonard Lake
Mary Lake
Beattie Lake
Menominee Lake
Big Stephen Lake
Rebecca Lake
Oxbow Lake
Little Hawk Lake
Maple Lake
Big Hawk Lake
Red Chalk Lake
Blue Chalk Lake
Plastic Lake
Kabakwa Lake
Harp Lake
Emsdale Lake
Dalrymple
Mississagua Lake



LAKESHORE CAPACITY STUDY
TROPHIC STATUS COMPONENT
Muskoka-Haliburton Area
0 5 10 15 20 25 30 35
Lake and stream water chemistry, plankton sampling, and hydrologic gauging
1976-1980
1979-1980
1981-1982
1982
Lake and stream water chemistry and plankton sampling
1976-1978
1976-1980
1979-1980
NOTE Symbol sizes for cartographic purposes only
Lake Huron
Toronto
Lake Ontario
Parry Sound
Huntsville
Bracebridge
Gravenhurst
Orillia
Minden
Haliburton
Bancroft
Georgian Bay
Lake Simcoe
Lake Couchiching
Lake Joseph
Rosseau Lake
Muskoka Lake
Muskoka Bay
Lake Vernon
Peninsula Lake
Fairy Lake
Lake of Bays
Mary Lake
Skeleton Lake
Doe Lake
Perch Lake
Oudaze Lake
Walker Lake
Jerry Lake
Harp Lake
Rebecca Lake
Solitaire Lake
Little Clear Lake
Buck Lake
Troutspawn Lake
Seventeen Mile Lake
McFadden Lake
Clinto Lake
Tock Lake
Paint Lake
Mink Lake
Chub Lake
Grandview Lake
Blue Chalk Lake
Shoe Lake
Black Lake
Dickie Lake
Heeney Lake
Red Chalk Lake
Crosson Lake
Pine Lake
Cinder Lake
Clear Lake
Bigwind Lake
Leonard Lake
Fawn Lake
Leech Lake
Wood Lake
Big Stephen Lake
Menominee Lake
Clearwater Lake
Kawagama Lake
Johnson Lake
Macdonald Lake
North Lake
Percy Lake
Redstone Lake
Eagle Lake
Moose Lake
Basshaunt Lake
Glen Lake
Halls Lake
Green Lake
Hurricane Lake
Two Islands Lake
Long Lake
Esson Lake
Kashagawigamog Lake
Mountain Lake
Twelve Mile Lake
Gull Lake
Little Bob Lake
Sludge Lake
Davis Lake
Head Lake
Four Mile Lake
Dalrymple Lake
Three Mile Lake
Kahshe Lake
Silent Lake
Mississagua Lake
Lower Buckhorn Lake
Lovesick Lake
Lake St. Peter
Glamour Lake
Billings Lake
Severn River
Black River
Head River
Burnt River
Irondale River
Muskoka River
THE DISTRICT OF PARRY SOUND
THE DISTRICT MUNICIPALITY OF MUSKOKA
THE DISTRICT OF NIPISSING
THE PROVISIONAL COUNTY OF HALIBURTON
THE COUNTY OF VICTORIA
THE COUNTY OF PETERBOROUGH
THE COUNTY OF SIMCOE
THE COUNTY OF HASTINGS

FOREWORD

Community planners and other professionals involved in the preparation of planning policies for lakeshore development and in the review of specific subdivision proposals have always found it difficult to determine objectively the impact of development on the natural environment. In response to this challenge, the Lakeshore Capacity Study was undertaken to provide a planning tool to assist in evaluating the effects of cottage development on inland lakes and lakeshores. Central to the task was the need to gain a clearer understanding of the relationship between cottage development and its impacts on selected aspects of the natural environment.

To accomplish these objectives, the Ministry of Municipal Affairs carried out the Lakeshore Capacity Study in cooperation with the Ministry of the Environment and the Ministry of Natural Resources.

The Muskoka-Haliburton area of central Ontario was chosen as the Study Area. The homogeneity of the area, which is part of one physiographic region, reduced the need to account for major natural variations among the lakes and watersheds. In addition, the extent of existing development on the lakes varied, permitting an examination of situations ranging from no development to full development.

The Study involved measuring the source of environmental impact, in terms of the lakeshore cottages and their use, and the impact of cottage development in terms of water quality, fisheries, and wildlife habitat. The methods of prediction derived from the research were linked in a simulation model, which is capable of predicting trends in the values of key indicators of impact.

As the research, analysis and findings of the Study are documented in a set of seven reports, selective reading may be desirable. Those readers who have a general interest in the work are advised to read the Committee Report first, as it provides an overall summary of the findings. This should be followed by the Integration report, in which the simulation model is described. Readers with more specialized interests will find the details of each component of the Study in the other five reports. In each report, the reader can select from the table of contents the most important chapters for his or her purposes.

The end product of the Study, the Ontario Lakeshore Capacity Simulation Model, has several features worth noting. The spatial unit addressed by the model is a single lake and the lakeshore. When the model is applied, the number of unknowns related to the natural environment can be substantially reduced, making it easier for planners or other professionals to weigh the environmental effects of development. The model goes a step further to permit predictions of the impact of cottage development when different management policies are selected.

The scope of the simulation model demands some explanation. In its present form, the model applies to cottage development on inland lakes in the Study Area, where the research was conducted. However, the methods of prediction can be adapted to other parts of the province, as long as differences in conditions are taken into account.

The purpose of the Study was to measure the environmental impact of cottages. Commercial and industrial uses were excluded deliberately, in order to simplify the difficult task of measuring cottage impact. The flexibility inherent in the simulation model makes it possible to add other types of land use later, if so desired.

Most of the existing cottage development in the Study Area is located in a single tier along the shoreline. For this reason, the simulation model applies to the immediate lakeshore and not to backshore development. Again, the methods of prediction developed for cottages near the lake can be adapted to measure the impact of cottage development in other forms.

The model was designed to measure the physical and chemical impacts of cottages. Accordingly, it does not address other planning concerns, such as social and economic impacts. While these were recognized as essential considerations in decision-making, the specific objective of Phase III of the Lakeshore Capacity Study was to find practical ways of producing better technical data regarding environmental impact.

Now that Phase III of the Study is completed, with the production of the Ontario Lakeshore Capacity Simulation Model (OLCSM), the next step envisaged is to apply the model experimentally within the government. In this setting, model output can be tested in a variety of actual development situations. When this period of experimental use has been concluded and the results assessed, the three participating ministries will be able to determine whether the model should be adapted to other parts of the province and whether it should be made available more widely.

The Ministry of Municipal Affairs considers the OLCSM to be a potential planning tool but recognizes that the technical and organizational implications of its use must be examined. While this is underway, the model will be available for testing as an additional planning tool to supplement the information normally required to evaluate a planning policy or development proposal. However, the model will not be used in the decision-making process, which will still rest on the customary range of planning considerations.

This Trophic Status report describes the improved methods of predicting water quality which were developed by scientists in the Ministry of the Environment, as part of Phase III of the Lakeshore Capacity Study.

ACKNOWLEDGMENTS

The authors wish to acknowledge the invaluable contributions to this study made by the Limnology Unit field staff – J. Findeis, R. Girard, J. Jones, B. Locke, L. Scott, and by the Ontario Ministry of the Environment Laboratory Services Branch staff, particularly F. Tomassini and C. Chun.

In its early stages, the study was reviewed by Drs. F. Rigler and J. Shapiro, both of whom provided excellent advice. The Lakeshore Capacity Study component managers, Al Burger, Jean Downing, Dave Euler and Al McCombie, assisted greatly in developing the initial study plan. Helpful comments on the summary report were provided by Drs. G. Nürnberg and V. Smith.

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GLOSSARY OF SYMBOLS

Symbol	Definition	Units*
MORPHOMETRY		
A_o	lake area	L^2
A_d	catchment or watershed area	L^2
V	lake volume	L^3
z	measured epilimnetic depth	L
$\bar{z}$	lake mean depth	L
z_{max}	lake maximum depth	L
D_L	development of lake length	—
D_V	development of lake volume	—
D_d	drainage density, stream length per unit drainage area	L^{-1}
D_T	depth of thermocline	L
F	fetch of lake	L
HYDROLOGY AND METEOROLOGY		
P	precipitation	L
R	surface runoff into lake	L
G	net ground water gain by the lake	L
O	outflow from the lake	L
E	net evaporation from the lake	L
LE	evaporative heat loss	Cal $L^{-2}T^{-1}$
r	net radiation	Cal $L^{-2}T^{-1}$
T	heat storage of the lake	Cal $L^{-2}T^{-1}$
B	sensible heat loss of the lake	Cal $L^{-2}T^{-1}$
q_s	areal water load	LT^{-1}
Q	lake outlet discharge, lake outflow volume	L^3T^{-1}
TVA	Tennessee Valley Authority	
D	distance	L
L	stream length	L
OR	organic deposits	%
ST	shallow till	%
E	total phosphorus export	$ML^{-2}T^{-1}$
τ_w	water replenishment time	T
MODELLING		
J_T	total phosphorus input from all sources	MT^{-1}
J_O	total phosphorus loss from the lake by outflow	MT^{-1}
J_A	anthropogenic total phosphorus input	MT^{-1}
J_{NAT}; J_N	total input of TP from natural sources	MT^{-1}
J_{PR}	total phosphorus input to lake from precipitation	MT^{-1}
L	areal loading rate of total phosphorus	$ML^{-2}T^{-1}$
L_N	annual natural total phosphorus load	$ML^{-2}T^{-1}$
L_A	maximum possible development TP load	$ML^{-2}T^{-1}$
L_T	maximum total phosphorus load	$ML^{-2}T^{-1}$
D	grain size measurement	L
H	predicted annual fish harvest	$ML^{-2}T^{-1}$
E_C	summer fishing effort by cottagers	angler hours
E_n	summer fishing effort by non-residents	angler hours
E_s	total summer fishing effort	angler hours
E_w	winter fishing effort	angler hours
N	number of dwellings	
MEI	morphoedaphic index	ML^{-4}
$\overline{B}$	lake average dry biomass of emergent Diptera	$ML^{-2}T^{-1}$
R_p	retention coefficient, fraction not lost via outflow	—
R_s	retention capacity	—
S	sedimentation rate measured by lake mass balance	$ML^{-2}T^{-1}$
T	whole-lake sedimentation rate estimate	$ML^{-2}T^{-1}$
SAR	sediment accumulation rate	$ML^{-2}T^{-1}$
SAR_s	site-specific sediment accumulation rate	$ML^{-2}T^{-1}$
V	sedimentation velocity	LT^{-1}
A_s	area of the plane through which sedimentation occurs	L^2
t	time	T
PB	phytoplankton biomass	ML^{-3}
CPB	corrected phytoplankton biomass	ML^{-3}
B_z	total crustacean biomass	ML^{-3}
B_h	herbivorous crustacean biomass	ML^{-3}
FR	zooplankton filtering rates	L^{-3}animal$^{-1}T^{-1}$
AHOD	areal hypolimnetic oxygen deficit	$ML^{-2}T^{-1}$
RT	response time	T
σ	sedimentation rate constant	T^{-1}
ϕ	sum of the loss rate constants $= Q/V + \sigma$	T^{-1}
SUBSCRIPTS		
SO	spring overturn	
SS	summer stratified	
FO	fall overturn	
IC	ice covered	
IF	ice free	
AN	annual	

EP, E	epilimnion
EM	epi- and metalimnion
H	hypolimnion
OUT	lake outlet
MEAS	measured
N	natural load
T	total load

STATISTICAL SYMBOLS

n	sample number
SE	standard error
r^2	square of correlation coefficient
p	probability
c.v.	coefficient of variation
ANOVA	analysis of variance test
t	value of the t-test

CHEMICAL SYMBOLS

H^+	hydrogen ion	μgL^{-1}
pH	inverse log of the hydrogen ion concentration	
P	phosphorus	μgL^{-1}
TP	total phosphorus	μgL^{-1}
DP	dissolved phosphorus	μgL^{-1}
N	nitrogen	μgL^{-1}
TN	total nitrogen	μgL^{-1}
TIN	total inorganic nitrogen	μgL^{-1}
TON	total organic nitrogen	μgL^{-1}
IN	inorganic nitrogren	μgL^{-1}
TKN-N	total Kjeldahl nitrogen	μgL^{-1}
NO_3-N	nitrite nitrogen	μgL^{-1}
NO_2-N	nitrate nitrogen	μgL^{-1}
NH_4-N	ammonia nitrogen	μgL^{-1}
Ca	calcium	μeqL^{-1}
Mg	magnesium	μeqL^{-1}
Na	sodium	μeqL^{-1}
K	potassium	μeqL^{-1}
Cl	chloride	μeqL^{-1}
SO_4	sulphate	μeqL^{-1}
DOC	dissolved organic carbon	mgL^{-1}
DIC	dissolved inorganic carbon	mgL^{-1}
TC	total carbon	mgL^{-1}
Fe	iron	mgm^{-1}
Pb	lead	mgm^{-3}
Mn	manganese	mgm^{-3}
Si	silica	mgL^{-1}
TDS	total dissolved solids	mgL^{-1}
Pb-210	radioactive lead	dpm
SD	secchi depth	m

* M = mass; T = time; L = length

I. INTRODUCTION

To assess current and future proposals for shoreline development of Ontario's inland lakes, scientific tools are needed to evaluate the effects of human use on the lakes and their watersheds. Quantitative rather than qualitative relationships between recreational, as well as non-recreational use and lake and watershed characteristics are required. Useful relationships can often be expressed in the form of predictive mathematical models.

The Lakeshore Capacity Study was initiated to provide such models for lakes in the Precambrian region of southern Ontario. The components of the aquatic and terrestrial systems that were identified as of prime concern to planners and managers responsible for evaluation of development proposals included the wildlife habitat of the terrestrial systems, the fisheries of the aquatic systems, microbially-related human health considerations of the aquatic systems, and the trophic status of the aquatic systems. This report summarizes the research studies carried out by the Limnology and Taxonomy Section (now the Aquatic Ecosystems Section) of the Ontario Ministry of the Environment with the purpose of assessing the effects of shoreline development on the trophic status of lakes.

A. TROPHIC STATUS

The trophic status of a lake is usually assessed by measurement of specific water quality parameters including chlorophyll *a* concentration (a surrogate measurement for phytoplankton biomass), water clarity (often as Secchi disc depth) and rate of loss of oxygen from profundal waters. It is well-known that phosphorus, the algal nutrient present in shortest supply, controls the trophic status of most Precambrian lakes because of its relationship with algal biomass and production. Linkages between total phosphorus content and trophic status formed the basis for the conceptual model developed by Dillon and Rigler (1975) for the assessment of the effects of lakeshore development on the trophic status of lakes. Although Dillon and Rigler (1975) developed predictive models, they were based on the untested assumption that shoreline development using septic systems for waste disposal provides phosphorus to the associated lakes. Many of the models were also developed on the basis of information collected from a large number of lakes located all over the world; lakes in Precambrian regions of Ontario typically exhibit a much smaller range in trophic status. Existing models cannot therefore provide

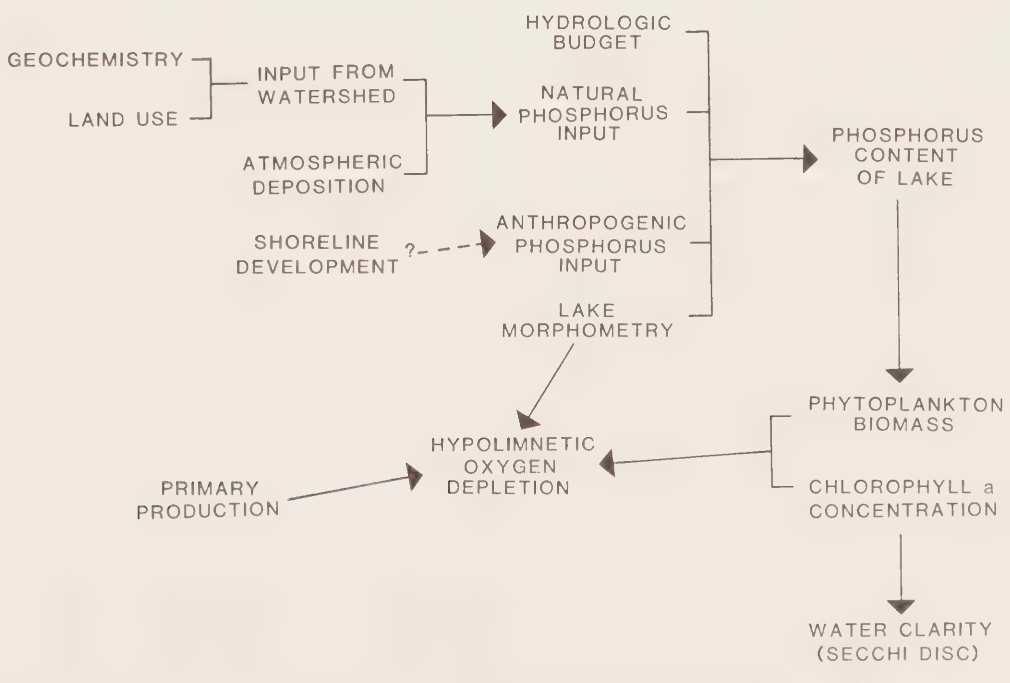


Figure 1 Conceptual model used in the Lakeshore Capacity Study of the relationships linking phosphorus and the trophic status of lakes.

accurate predictions with an acceptable degree of confidence over this smaller range in trophic conditions.

The conceptual model that formed the basis for the trophic status component of the Lakeshore Capacity Study is outlined in Figure 1. The potential effects of shoreline development on trophic status were considered to result from the use of septic systems. Other anthropogenic activities in, on and around lakes were considered to bear little relationship to trophic status.

Other anthropogenic stresses unrelated to shoreline development may affect the water quality of Precambrian lakes. Acidic deposition, a result of the oxidation and hydrolysis of sulphur and nitrogen oxides released as a product of fossil fuel combustion or the smelting of sulphide ores, may be the single most significant environmental problem in eastern North America. Lakes on the Precambrian Shield are particularly sensitive to the deposition of these strong acids. As loading rates of acid in the study area are as high as those in any area in the world, effects on the aquatic biota are to be anticipated (Dillon et al. 1978). In the latter sections of this report, the effects of acidic deposition on trophic status variables are analyzed to ascertain whether nutrient-trophic status relationships are influenced by acidic deposition.

B. STUDY DESIGN

Several approaches were adopted to quantify the effects of lakeshore development on trophic status.

In the first approach, a set of six lakes which, according to existing trophic status models (Dillon and Rigler 1975) exhibited the maximum range in potential significance of shoreline development was selected for study. These six lakes were termed the "A" lakes, and were located in the District Municipality of Muskoka or in Haliburton County.

The potential significance of shoreline development was expressed as the ratio of the estimated potential anthropogenic TP input to the lake (i.e., the potential input from shoreline development) and the estimated total TP input to the lake. These ratios ranged from 0 for Jerry Lake (no shoreline development) to 0.61 for Harp Lake (Table 1). The readily measureable components of the TP mass balances of these lakes were subsequently measured as accurately as possible for four years, as was the hydrologic budget of each lake. Existing trophic status models were calibrated using the subset of the "A" lakes (Jerry and Red Chalk Lakes) with no (or almost no) shoreline development or anthropogenic TP inputs, then applied to the developed lakes. Differences between measured TP concentrations in the developed lakes and levels predicted on the basis of models calibrated using the undeveloped lakes were assumed to be attributable to anthropogenic TP inputs, which in turn, were quantitatively related to the extent of shoreline development on each lake. This approach is described in detail in Section III.

Table 1. Estimated ratio of anthropogenic (J_A) to total phosphorus input (J_T) for the "A" lakes.

Lake	J_A/J_T
Harp	0.61
Dickie	0.54
Blue Chalk	0.27
Chub	0.26
Red Chalk	0.04
Jerry	0.0

The second approach utilized to assess the effects of lakeshore development on trophic status was to compare the trophic status of selected lakes before and after shoreline or other types of development occurred. No such historical comparison exists for any Precambrian lakes in Ontario. Nine lakes ("B" lakes) were selected. The anticipated changes in development on these lakes are outlined in Table 2. However, few of these changes actually occurred over the course of the Lakeshore Capacity Study. Because of this, monitoring of these lakes by the Ontario Ministry of the Environment has continued (1980-83) and will continue until the proposed developments are carried out or abandoned. Long-term changes in the "B" lakes are not described in this report.

Nevertheless, data collected for the "B" lakes as part of this study were very useful. Information collected for the "A" lakes and their watersheds was used to develop new models linking individual components of the overall conceptual model. Many of these relationships were then tested on the independent data set collected for the "B" lakes. Models were developed for prediction of TP export (or yield) from terrestrial watersheds, for prediction of phytoplankton biomass (and chlorophyll *a* concentration) on the basis of nutrient status, for prediction of water clarity and hypolimnetic oxygen depletion rates, and for other linkages in the conceptual model. Several of these relationships depended on the geology and geochemistry of the terrestrial watersheds of the lakes. Because of this, a set of nine additional terrestrial watersheds was studied to extend the range in geological characteristics beyond that observed for the "A" and "B" lakes. These additional watersheds were termed the "export" watersheds.

Model development is described in detail in Section IV.

Table 2. Summary of development status for the "B" lakes. Number of development units at the start of the study, status in 1978, and expected future numbers are indicated.

Lake	Number of development units at start of study (1976)	Number of development units in 1978 (1978)	Expected number of of development units
Basshaunt	8	11	31
Glen	5	7	17
Buck	0	0	194
Little Clear	0	0	158
Solitaire	16	14	14
Walker	23	34	73
Bigwind	16	18	100
Crosson	1[1]	1[1]	0[2]
Gullfeather	0	0	0[2]

[1] Ontario Ministry of Natural Resources seasonal camp.
[2] within the boundaries of proposed provincial park and campsite.

II. DESCRIPTION OF THE STUDY AREA AND STUDY LAKES

A. LOCATION

The location of the study lakes, additional ("export") watersheds, and atmospheric deposition stations are indicated in Figure 2. All lakes are located in Haliburton County or in the District Municipality of Muskoka in Ontario. The nearest major urban centres are quite distant from the study area. Toronto and Peterborough are located roughly 200 and 120 km, respectively to the south. North Bay and Sudbury are 130 and 200 km, respectively to the north and northwest.

B. CLIMATE AND VEGETATION

Thirty-year averages of meteorological data taken from the Hydrological Atlas of Canada (1978) indicate that annual precipitation depth averages 90-110 cm in the Dorset area, the location of the field laboratory in the middle of the study area. A total of 240-300 cm of snow falls each year, generally between December 1 and April 10. January temperatures average –10°C, while in July temperatures average 17.5°C. Mean annual temperature is about 5°C. Lakes in the area generally are frozen from the first week of December to mid-April.

The study area is situated in the Great Lakes-St. Lawrence forest region, a region characterized by eastern white pine, red pine, eastern hemlock and yellow birch. Beech, white oak, sugar maple, basswood, eastern white cedar, red maple, red oak, white birch and poplar are also common in the area. Generally the forest stands on north-facing slopes of the lakes are coniferous, whereas south-facing slopes are deciduous.

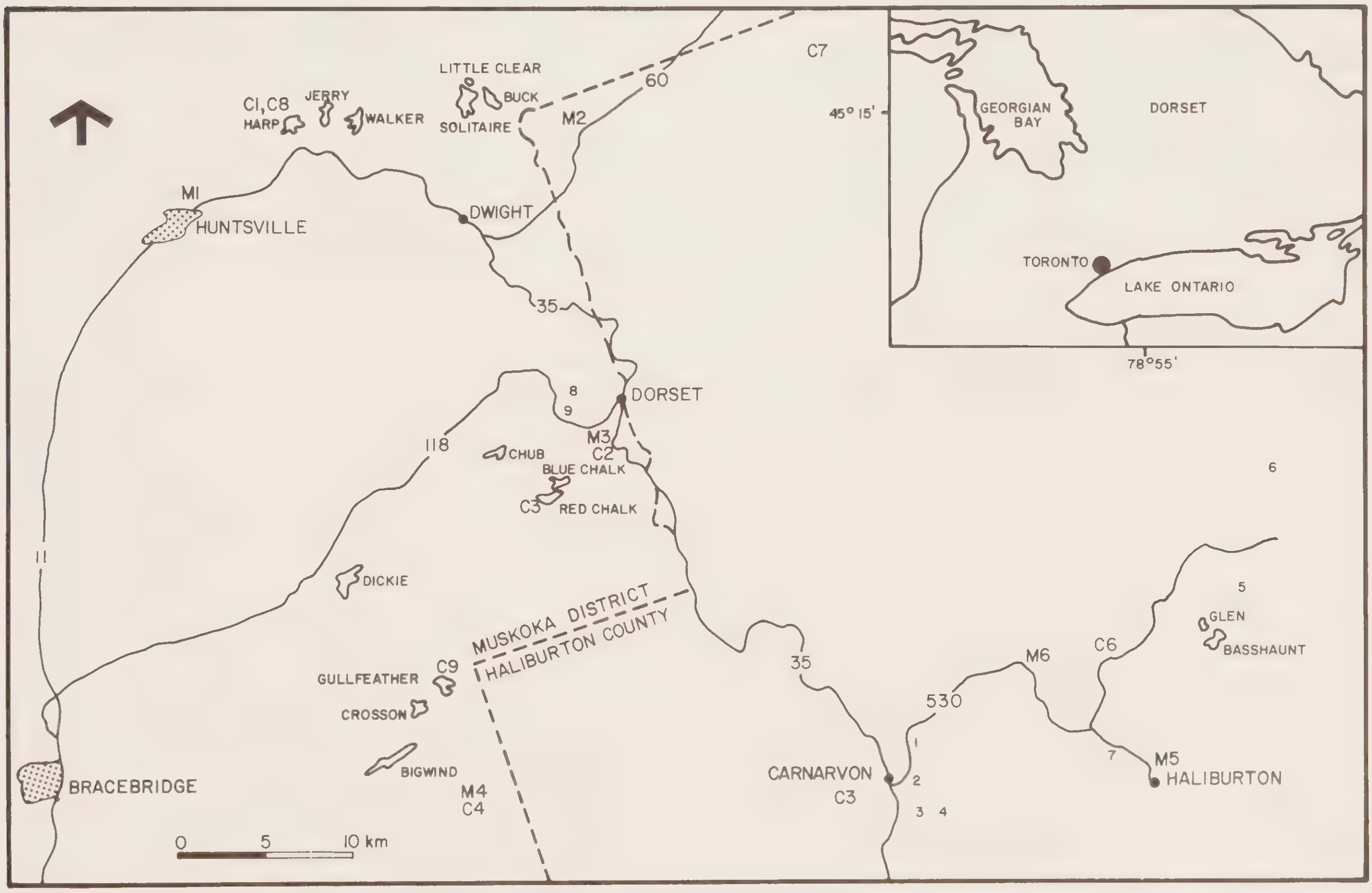


Figure 2 Map of the Lakeshore Capacity Study Area in Muskoka-Haliburton locating the study lakes, meteorological stations (M), bulk precipitation stations (C1 – C9) and export streams (1-Beech 1, 2-Twelve Mile North, 3-Twelve Mile South, 4-Duck 1, 5-Moose 1, 6-Haliburton 12, 7-Head 1, 8-Trading Bay 1, 9-Paint 1).

C. MORPHOMETRY

Detailed bathymetric tables and maps of the Lakeshore Capacity Study lakes are provided in a separate report (Nicolls et al. 1983). A summary of selected morphometric information is provided in Table 3. The lakes have morphometries typical of that for small shield lakes (Brunskill and Schindler 1971). They vary in area from 11 to 111 ha, in mean depth from 5 to 14.2 m and in maximum depth from 12 to 40 m. The ratio of (Ad + Ao)/V, a good correlate with lake productivity, ranges from 0.3 to 6.5, again typical of ranges observed for Shield lakes (Schindler 1971).

With the exception of Little Clear Lake, all the lakes have a single, well-developed outflow. Blue Chalk Lake drains into Red Chalk Lake, and Solitaire Lake into Little Clear Lake. Basshaunt, Crosson and Gullfeather Lakes each have small lakes draining into them. All other lakes are headwater lakes with many permanent and ephemeral streams and perhaps small ponds in their watersheds.

D. GEOLOGY

The study lakes and streams are situated on the Canadian Shield, an expanse of Precambrian bedrock covering almost one half of Canada. The Canadian Shield is one of the most seismically inactive areas in the world. The last orogeny (period of mountain building) of the Precambrian period was the Grenville orogeny which formed the Grenville province, a roughly rectangular strip of land 400 km wide and 2,000 km long stretching from Georgian Bay northeast to Labrador and underlying the study area. The bedrock of the Grenville province is of granitic composition, usually granitic gneisses and migmatites with marbles, quartzites, amphibolites and various igneous intrusives (pegmatites, diorites and metabasics) of less importance.

On the scale of individual lake watersheds, bedrock and surficial geology may be variable. As the minerology of the bedrock and surficial materials may influence nutrient fluxes, both the surficial and bedrock geology of the study lakes were mapped by air photo analysis and direct field observation. These analyses are interpreted in detail by Jeffries and Snyder (1983) and are summarized below.

Relative contribution to total watershed area of several bedrock and surficial features are listed in Table 4 for the "A" lakes and Table 5 for the "B" lakes. Gneiss sometimes found with thin beds of shist is the dominant bedrock in ten of the study lakes. Migmatite replaces gneiss in five other lakes. The bedrock of Glen Lake is distinct, composed of marble in the northern section contacting gneiss in the southern portion of the watershed.

During the Pleistocene period, four glacial advances, including the most recent or Wisconsin glaciation, have rounded and polished the rock ridges and knobs of the Grenville province, lowering the relief, scouring many river and lake basins and redepositing many types of glacial debris, including moraines, eskers, outwash sands and gravels and lake sediments. The surficial deposits and many characteristics of the watersheds of the lakes are attributable to this glacial activity.

Minor till plains (continuous moraine deposits >1m thick) and thin till deposits (<1 m thick) interrupted by rock ridges are the dominant surficial geological groups. Where thin tills are predominant, such as in Dickie, Crosson and Gullfeather Lakes, peat deposits are generally extensive (Tables 4 and 5). The surficial deposits of Glen Lake are unique among the study lakes. The north end of the watershed has extensive carbonate tills.

The watersheds of Harp, Jerry, Red Chalk and Blue

Table 3. Summary of selected morphometric attributes of study lakes. Parameters are lake area (A_o), volume (V), mean depth ($\bar{z}$), maximum depth (z_{maxd}), shoreline length (L), development of length (D_L) and volume (D_V) and watershed area excluding A_o (A_D). Notation and methods of calculation after Hutchinson (1957). Details in Nicolls and Reid (1983).

Lake	A_o (ha)	V ($m^3 \times 10^5$)	$\bar{z}$ (m)	z_{max} (m)	L (km)	D_L[1]	D_V[2]	A_d (ha)	$\frac{A_d + A_o}{V}$ (m^{-1})
Basshaunt	47.3	36.6	7.7	24	4.9	1.99	0.96	791	2.29
Bigwind	111.0	118.0	10.7	32	8.2	2.21	1.00	396	0.43
Blue Chalk	49.4	42.1	8.5	23	4.6	1.85	1.11	128	0.42
Buck	40.3	43.9	10.9	30	3.6	1.58	1.09	251	0.66
Chub	32.2	28.5	8.9	27	3.9	1.92	0.99	286	1.12
Crosson	56.8	47.7	8.4	25	3.9	1.45	1.01	501	1.17
Dickie	93.2	46.4	5.0	12	7.8	2.29	1.25	449	1.17
Glen	16.3	11.8	7.2	15	1.8	1.28	1.44	97	0.96
Gullfeather	65.9	31.5	4.8	13	5.3	1.83	1.11	982	3.33
Harp	66.9	82.6	12.4	40	4.6	1.57	0.93	509	0.70
Jerry	50.1	61.9	12.4	35	4.6	1.83	1.06	861	1.47
Little Clear	10.9	8.86	8.1	25	1.5	1.26	0.97	569	6.54
Red Chalk	56.9	80.8	14.2	38	4.8	1.80	1.12	611	0.83
Solitaire	124.0	164.01	13.3	31	6.0	1.51	1.29	379	0.31
Walker	68.2	42.1	6.2	17	6.4	2.20	1.09	258	0.77

[1] Development of length is defined as the ratio of length of shoreline to the length of the circumference of a circle of area equal to that of the lake.

[2] Development of volume is defined as the ratio of lake volume to that of a cone of basal area A_o and height z_{max}.

Chalk Lakes were partially submerged under glacial Lake Algonquin (about 10,000 years ago) and received sediment laden glacial meltwater. Jerry and Harp Lakes have substantial sand deposits and Red and Blue Chalk Lakes have outwash deposits in consequence (Table 4). Little Clear Lake was intersected by a regional fault which acted as a major glacial meltwater spillway. The largest deposit of sand and gravel is therefore located in the watershed of this lake (Table 5).

E. METHODOLOGY

The methodologies employed in carrying out the trophic status studies are described in detail by Scheider et al. (1983).

Precipitation was collected at up to 9 locations (Figure 2) in the study area between May 1976 and June 1980. Details of collection frequency, periods of collector operation, methods of pooling station data and calculation of elemental deposition are presented by Scheider et al. (1983).

Water samples were collected from all inlet and outlet streams of the study lakes, and from the extra ("export") streams (Figure 2). Stream stage or level was continuously recorded for all "A" lake inflows and outflows and converted to discharge using stage-discharge curves developed for each stage control structure (weir and/or flume). Details of control structure design, stage-discharge relationships, gauging and sampling frequency of the streams, manipulation of stream data, calculation of discharge from ungauged subwatersheds are provided by Scheider et al. (1983).

Sediment cores were collected from each of the "A" lakes. Methods of collection and chemical analysis are described in detail by Dillon and Evans (1982) and Evans et al. (1983).

All "A" and "B" study lakes were sampled at a single mid-lake station located at the deepest spot in each lake. A station was located at each of the two distinct basins in Red Chalk Lake. Chemical samples were collected at weekly intervals for the "A" lakes and at monthly intervals for the "B" lakes and were volume-weighted composites of all depths when the lake was isothermal. Separate volume-weighted composites were taken from epi-, meta- and hypolimnia when the lakes were stratified. Unlike all other samples, phytoplankton and chlorophyll *a* samples were collected through the euphotic zone, estimated as twice the Secchi disc depth. Lake samples were filtered through 250 μm mesh to remove coarse particulate material. Details of methods and frequency of sampling and of sample preservation are provided by Scheider et al. (1983).

Analytical methods for chemical analysis of precipitation, lake and stream water and lake sediments are provided by Ontario Ministry of the Environment (1981). Phytoplankton samples were counted by the Utermohl technique.

Table 4. Percent of area of watershed of 6 "A" lakes formed by 4 bedrock types and 7 surficial categories.[1]

Lake	Percent of Bedrock Type				Percent of Surficial Type						
	Gneiss	Diorite	Amphibolite and Schist	Migmatite	Minor Till Plain	Thin Till and Rock Ridges	Peat	Bedrock	Outwash	Sand	Pond
Blue Chalk	100	–	–	–	76.7	16.5	–	1.4	5.4	–	–
Chub	100	–	–	–	32.3	61.7	4.4	1.6	–	–	–
Dickie	–	–	–	100	4.7	81.4	13.9	–	–	–	–
Harp	68.5	3.1	28.4	–	49.9	38.5	6.3	0.3	–	4.3	0.8
Jerry	99.5	0.5	–	–	19.8	72.2	4.2	0.1	–	3.4	0.3
Red Chalk	100	–	–	–	57.8	26.5	3.4	2.0	6.9	–	3.6

[1] Categories are described and watershed characteristics analysed by Jeffries and Snyder (1983).

Table 5. Percent of area of watershed of 9 "B" lakes formed by 5 bedrock types and 8 surficial categories.[1]

Lake	Percent of Bedrock Type					Percent of Surficial Type							
	Marble	Amphibolite	Gneiss and Schist	Gneiss and Skarn	Migmatite and Granite Intrusives	Till Carbonate	Minor Till Plain	Thin Till and Rock Ridges	Peat Sand	Peat Till	Outwash	Sand	Pond
Basshaunt	–	–	100	–	–	–	56.5	28.4	3.0	1.6	2.9	–	7.6
Bigwind	–	–	–	–	100	–	73.4	21.1	4.4	1.2	–	–	–
Buck	–	–	100	–	–	–	86.6	10.5	–	2.4	–	–	0.5
Crosson	–	0.2	–	–	100	–	20.0	66.4	7.4	–	–	–	6.2
Glen	22.7	–	–	77.3	–	99.9	–	–	–	–	–	0.1	–
Gullfeather	–	0.2	–	–	99.8	–	33.4	50.4	9.3	0.6	–	0.9	5.4
Little Clear	–	4.4	95.6	–	–	–	75.2	–	–	–	24.8	–	–
Solitaire	–	0.3	99.7	–	–	–	81.2	11.4	–	–	6.7	–	–
Walker	–	0.3	99.7	–	–	–	90.3	8.0	–	1.2	–	0.5	–

[1] Categories are described and watershed characteristics discussed in Jeffries and Snyder (1983).

F. LAKE CHEMISTRY

Chemical attributes of the study lakes are summarized in Tables 6 through 9. Excluding Glen Lake, pH, conductivity, major ion and nutrient levels are typical of those observed in dilute, nutrient-poor Canadian Shield lakes (Armstrong and Schindler 1971). The geological setting of the lakes was the major determinant of major ion chemistry and colour of the lakes. Hence, Glen Lake, the only lake with extensive carbonate tills in its drainage basin had much higher alkalinity, pH, conductivity, DIC and TC than all other lakes (Tables 7 and 9). Chub, Dickie, Crosson and Gullfeather Lakes had pH levels of 5.6. – 5.9 while all other lakes had pH >6.2. These four lakes also had higher colour and lower Secchi transparency than all other lakes, probably because the watersheds of these four lakes had extensive deposits of thin till and peat (Table 3).

Average concentrations of total phosphorus (TP) over the four-year period during which data were collected ranged from 5.7 to 12.8 mg m^{-3} among the "A" lakes and 6.4 to 20.7 mg m^{-3} among the "B" lakes (Tables 8 and 9). Levels of total nitrogen (TN = TKN-N + NO_3-N + NO_2-N) ranged from 227 to 394 mg m^{-3} among the "A" lakes and from 296 to 504 mg m^{-3} among the "B" lakes. TN to TP ratios averaged 39:1 (range 28-53) by weight for the "A" lakes and 41:1 (range 24-52) among the "B" lakes. The magnitude of the ratio indicates that phosphorus is the element present in shortest supply for the primary producers in the lakes (Schindler 1977).

Lakes are often separated into five trophic categories, qualitative expressions of the degree of nutrient enrichment and biological response of the lakes. The categories are ultra-oligotrophic, oligotrophic, mesotrophic, eutrophic and hypertrophic. Often only the three middle categories are employed. The assignment of lakes to particular trophic categories is subjective as clear boundaries between the categories are difficult to identify.

Vollenweider and Kerekes (1980) state that the categorization is not simply an academic exercise despite these limitations as there is a need for limnologists to employ a mutually comprehensible language with each other and with lake managers and planners. Vollenweider and Kerekes (1980) developed a probabilistic approach to trophic designation. They indicate that lakes with TP levels of $\simeq$15 mg m^{-3} have an equal probability ($\simeq$45%) of being oligotrophic or mesotrophic. A smaller fraction of such lakes may even be ultra-oligotrophic or eutrophic. Using this equivalence concentration for TP to assign the Lakeshore Capacity Study lakes to trophic categories, Glen Lake (TP = 20.7 mg m^{-3} would be mesotrophic. All other lakes have <15 mg m^{-3} of TP and should be categorized as oligotrophic. Using other trophic state indicators (chlorophyll *a*, Secchi depth) would give slightly different categorizations.

Table 6. Summary of chemistry of the "A" lakes. Results are the mean of 4 annual means, (June 1, 1976 – May 31, 1980) of volume-weighted, composite samples.

Lake	Conductivity (μmho cm^{-2})	Colour (Hazen Units)	Cations[1] (μeq L^{-1})	Anions[2] (μeq L^{-1})	pH	Alkalinity[3] (μeq L^{-1})
Blue Chalk	30.3	5.2	253	171	6.47	103
Chub	31.5	18.3	250	214	5.58	62.2
Dickie	29.9	15.2	235	207	5.67	58.6
Harp	36.1	6.9	302	215	6.22	101
Jerry	38.3	10.3	328	231	6.19	104
Red Chalk Main	30.6	5.3	260	173	6.24	103
East	33.0	7.9	276	177	6.36	126

[1] $Ca^{2+} + Mg^{2+} + Na^{+} + K^{+}$
[2] $Cl^{-} + SO_4^{2-}$
[3] Corrected fixed-endpoint alkalinity

Table 7. Summary of chemistry of the "B" lakes. Values are the mean of 4 annual means, (June 1, 1976 – May 31, 1980) of volume-weighted, composite samples.

Lake	Conductivity (μmho cm^{-2})	Colour (Hazen Units)	Cations[1] (μeq L^{-1})	Anions[2] (μeq L^{-1})	pH	Alkalinity[3] (μeq L^{-1})
Basshaunt	46.1	7.0	367	203	6.61	149
Bigwind	31.7	5.8	264	188	6.22	83.4
Buck	33.7	5.5	288	184	6.35	112
Crosson	27.8	12.8	219	187	5.65	58.4
Glen	169.0	5.4	1,740	602	7.23	1,200
Gullfeather	31.1	24.4	255	206	5.84	79.4
Little Clear	39.5	10.7	331	188	6.49	202
Solitaire	32.6	5.5	272	179	6.42	134
Walker	35.7	5.8	303	208	6.43	107

[1] $Ca^{2+} + Mg^{2+} + Na^{+} + K^{+}$
[2] $Cl^{-} + SO_4^{2-}$
[3] Corrected fixed-endpoint alkalinity

Table 8. Summary of the nutrient chemistry of the "A" lakes. Values are the mean of 4 annual means (June 1, 1976 to May 30, 1980). DIC is the mean of two annual values, 1978 to 1980. Samples used to generate means were volume-weighted, composite samples.

Lake	TP	TKN-N	NO_3^--N	NH_4^+-N	DOC	DIC	TC	Fe	Mn	Si	Secchi Depth
	(mg m^{-3})				(mg L^{-1})			(mg m^{-3})		(mg L^{-1})	(m)
Blue Chalk	7.07	210	17.5	22.7	3.07	1.03	5.09	80.5	36.7	0.31	6.79
Chub	12.20	296	63.7	33.6	6.19	0.77	8.06	386.0	53.0	1.29	3.29
Dickie	12.80	325	41.1	36.1	6.68	0.59	7.77	242.0	49.3	0.73	2.79
Harp	7.80	245	76.5	13.8	4.80	1.11	6.83	148.0	64.1	1.36	3.90
Jerry	9.14	256	138.0	21.8	5.54	1.26	8.15	237.0	77.1	1.92	3.89
Red Chalk Main	5.71	219	87.7	17.2	3.64	1.25	5.91	128.0	40.6	1.06	6.10
East	7.40	301	43.1	72.3	4.60	1.59	7.81	364.0	64.5	1.25	5.10

Table 9. Summary of the nutrient chemistry of the "B" lakes. Values are the mean of 4 annual means (June 1, 1976 to May 30, 1980) except for DIC which is the mean of two annual values 1978-1980 and TC which is calculated from 1976-1979. Samples used to generate means were volume weighted composite samples.

Lake	TP	TKN-N	NO_3^--N	NH_4^+-N	DOC	DIC	TC	Fe	Mn	Si	Secchi Depth
	(mg m^{-3})				(mg L^{-1})			(mg m^{-3})		(mg L^{-1})	(m)
Basshaunt	8.05	263	90.8	26.1	4.47	1.88	7.44	85.8	45.1	1.06	4.68
Bigwind	7.75	237	100.0	19.6	3.83	0.96	5.50	97.3	28.8	0.72	4.99
Buck	7.10	259	66.3	27.8	3.58	1.24	5.88	130.0	63.8	0.65	6.05
Crosson	11.70	289	102.0	28.0	5.16	0.56	6.20	330.0	39.6	0.93	3.66
Glen	20.70	464	60.3	158.0	3.94	14.60	20.40	31.5	59.3	1.97	4.69
Gullfeather	13.40	362	126.0	58.4	6.87	0.77	7.51	462.0	69.8	1.24	2.49
Little Clear	11.20	411	64.4	124.0	4.24	2.01	7.93	605.0	132.0	1.31	5.20
Solitaire	6.40	229	67.0	19.0	2.79	1.17	5.30	48.1	16.5	0.24	7.64
Walker	6.83	309	46.6	34.1	4.56	1.01	6.27	68.2	64.9	0.48	5.55

III. EVALUATION OF THE EFFECTS OF LAKESHORE DEVELOPMENT ON THE TROPHIC STATUS OF THE STUDY LAKES

A. INTRODUCTION

The importance of nutrients, particularly phosphorus, in governing the trophic status of lakes was emphasized in Section I of this report. In this section, we attempt to evaluate the effects of lakeshore development on the trophic status of the study lakes.

The influence of development is very difficult to measure directly because the capability of natural soil deposits around lakes to assimilate or retain phosphorus discharged by septic tile beds is extremely variable. Extrapolations to entire lakes from a few study sites cannot be made with confidence.

Two indirect approaches have therefore been utilized
a) the nutrient regimes of the developed and undeveloped lakes are compared,
b) several biological characteristics of the developed and undeveloped lakes that are considered to be dependent on trophic status are compared.

Of the six intensively studied ("A") lakes, two (Jerry and Red Chalk) had very little or no development on their shoreline or in their watersheds, while the other four had significant quantities of development (see Table 1). However, direct comparison of the phosphorus (or other nutrient) content of the developed and undeveloped lakes is not useful for elucidation of the effects of development because of the natural variability in nutrient content in undeveloped lakes. On the other hand, the conceptual links between lakeshore development and a lake's phosphorus budget are well-established. It is therefore possible to formulate the conceptual model in mathematical terms, calibrate it using the undeveloped lakes, then utilize it to ascertain the effects of lakeshore development on the TP budget, and hence, on the trophic status of the four developed lakes.

The second approach employed is the direct comparison of several biological components of the developed and undeveloped lakes. Phytoplankton, zooplankton and macrophytes are considered. However, natural differences in trophic status confound the interpretation of community data, restricting the usefulness of this approach. On the other hand, direct comparison does allow implicit evaluation of the influence of lakeshore development on biological communities that are not related to changes in nutrient supply.

B. THE PHOSPHORUS BUDGET APPROACH

1. BACKGROUND

The factors that determine the total phosphorus (TP) concentration in lakes include lake morphometry, the hydrologic budget, the TP input or loading rate, and the TP loss rate. The conceptual model that links these factors was outlined in Figure 1. The morphometries of the "A" lakes were measured at the beginning of this study. The hydrologic budgets of the "A" lakes were directly measured for the period June 1976-May 1980. The total phosphorus inputs from all significant sources other than those associated with shoreline development were also measured for the same period; these sources include atmospheric deposition and watershed runoff. Loss of TP by outflow from each lake was measured for the duration of the study. Loss of TP by sedimentation within each lake must also be quantified, but the rate of accumulation of TP in the sediments of lakes is very difficult to measure independently. The sedimentation rate of TP can be calculated by difference between the total input rate and the loss rate attributed to outflow if corrections for change in TP concentration in the lake (if any) are made. This methodology is only suitable for the undeveloped lakes since the actual total input is unknown for the developed lakes. Development of a general sedimentation rate model or independent measurement of sedimentation rate is therefore necessary.

The individual components of the hydrologic and TP budgets of the "A" lakes are summarized in the subsequent sections. Because some substances, notably several of the major ions present in freshwaters, behave conservatively with respect to lakes (that is, they are removed from the aquatic system only by outflow but not by biological or chemical processes resulting in sedimentation), their mass balances are employed to validate the hydrologic budgets.

2. HYDROLOGY OF THE "A" LAKES

The components of the hydrologic cycle are related by the water balance equation, an expression of the principle of the conservation of mass. For a lake, this equation can be written as:

$$P + R + G - O - E = \Delta V \qquad (1)$$

Where: P = precipitation to the lake surface
R = surface runoff into the lake
G = net ground water gain by the lake
O = outflow from the lake
E = net evaporation from the lake
ΔV = change in lake storage, or volume.

Water balances were constructed for four consecutive 12-month periods beginning in June 1976 and for the entire 1976-80 period. The 12-month periods from June to May were chosen as the standard water year for several reasons. Hydrological data collected over the period June 1976 – May 1980 was relatively complete. Mass balance models have been historically used to predict TP concentration in lakes at spring overturn, which commonly occurs in May in the study lakes. Selection of the June – May water year minimizes changes in the water balance resulting from changes in storage since the snowpack is gone and the spring runoff has ended by June 1.

Because each term in the water balance equation except groundwater was measured or estimated independently, it is possible to evaluate the overall accuracy of the hydrologic budgets. This evaluation is performed after each term in the water balance equation is considered.

i) PRECIPITATION

Precipitation depth was measured at six meteorological stations (Figure 2) within the study area. Annual precipitation depth at these locations is summarized in Table 10. Precipitation depth at each lake or calibrated watershed was taken from the nearest station. Mean annual values for the four-year study period ranged from 0.96 m yr^{-1} to 1.17 m yr^{-1}. Long-term mean annual precipitation depth for the area is 0.9 – 1.1 m yr^{-1}, declining from west to east (Hydrological Atlas of Canada 1978).

There was no significant difference in the annual precipitation depth among the seven stations (using 2-way ANOVA at 0.05 level of significance). Repeating the analysis on a monthly basis, significant differences in precipitation depth among the stations occurred only in February and March. However, on a daily basis considerable variability occurred in some instances. Spatial variability in precipitation depth among five collectors located on a single lake (Red Chalk Lake) was not significant (Jeffries et al. 1978). Significantly more precipitation fell on the open lake than under the surrounding forest canopy due to interception by the canopy.

The volume of precipitation that fell on each lake and the relative importance of this component in the water balance are summarized in Tables 12-17.

ii) RUNOFF

Annual runoff (annual streamflow/watershed area) for each of the 36 gauged streams is summarized in Table 11. Mean annual values were 0.34 m yr^{-1}; 0.43 m yr^{-1}; 0.54 m yr^{-1} and 0.56 m yr^{-1} in 1976-77, 1977-78, 1978-79 and 1979-80 respectively, but watershed-to-watershed variability was quite large. The overall 4-year mean runoff was 0.48 m yr^{-1}, comparing well with the long-term mean annual range in runoff (0.40 – 0.50 m yr^{-1}) in the study area (Hydrological Atlas of Canada 1978). Yield (runoff/precipitation depth) standardizes the runoff data for variations in precipitation depth. Mean annual values of yield for the 36 gauged watersheds were 0.40, 0.49, 0.45 and 0.51 in 1976-77, 1977-78, 1978-79 and 1979-80 respectively.

A detailed discussion of spatial and temporal variability in streamflow will be provided elsewhere as it is beyond the scope of this report. Explanations for the observed differences between watersheds in runoff have also been investigated (Scheider unpub.). Watersheds were grouped based on physical and geochemical characteristics (Jeffries and Snyder 1983), and ANOVA used to assess between-group variance in streamflow indices including annual runoff. Runoff was significantly ($p < 0.05$) greater from watersheds with surficial geology consisting of >3% organic deposits than from those with <3% organic deposits (mean runoff values 0.50 m yr^{-1} and 0.43 m yr^{-1} respectively).

Results of stepwise regression using eleven streamflow indices as dependent variables and fifteen watershed indices as independent variables and the remaining ANOVA's on grouped streamflow data will also be described in detail elsewhere.

Table 10. Annual precipitation depth (m) for the study streams and lakes for the 1976-77, 1977-78, 1978-79 and 1979-80 periods.

		Annual precipitation depth (m)				
Station	Lake or Watershed	1976-77	1977-78	1978-79	1979-80	Mean
M1	Harp, Jerry, Walker	0.904	0.906	1.14	0.941	0.973
M2	Buck, Little Clear, Solitaire	0.952	1.23	1.20	1.15	1.13
M3[1]	Red Chalk, Blue Chalk, Dickie, Chub, Paint Lake #1, Trading Bay #1	0.823	0.818	1.35	1.18	1.04
M4	Bigwind, Gullfeather, Crosson	1.26	0.791	1.07	1.18	1.08
M5	Basshaunt, Glen, Moose #1, Head #1, Haliburton #12	0.809	0.961	1.01	1.04	0.955
M6	12 Mile N., 12 Mile S., Duck #1, Beech #1	1.10	1.14	1.13	1.31	1.17

[1] Precipitation for Dickie Lake was 0.849 m in 1976-77 and 0.806 m in 1977-78.

Table 11. Annual runoff (m yr^{-1}) for 36 gauged streams for 1976-77, 1977-78, 1978-79 and 1979-80 periods.

Watershed		1976-77	1977-78	1978-79	1979-80	Mean
Blue Chalk	1	0.316	0.196	0.231	0.397	0.285
	outlet	0.364	0.403	0.544	0.581	0.473
Chub	1	0.260	0.273	0.363	0.329	0.306
	2	0.371	0.533	0.523	0.626	0.513
	outlet	0.271	0.427	0.515	0.567	0.445
Dickie	5	0.510	0.462	0.552	0.657	0.545
	6	0.388	0.464	0.594	0.569	0.504
	8	0.261	0.322	0.305	0.561	0.362
	10	0.492	0.500	0.638	0.691	0.580
	11	0.298	0.388	0.502	0.502	0.423
	outlet	0.351	0.407	0.496	0.537	0.448
Harp	3	0.467	0.503	0.639	0.636	0.561
	3A	0.262	0.588	0.630	0.530	0.503
	4	0.330	0.411	0.656	0.555	0.488
	5	0.422	0.415	0.553	0.692	0.521
	6	0.201	0.263	0.382	0.324	0.293
	6A	0.275	0.334	0.439	0.458	0.377
	outlet	0.307	0.440	0.617	0.591	0.489
Jerry	1	0.314	0.418	0.527	0.507	0.442
	3	0.400	0.412	0.570	0.549	0.483
	4	0.296	0.343	0.490	0.511	0.410
	outlet	0.394	0.422	0.607	0.490	0.478
Red Chalk	1	0.413	0.443	0.555	0.563	0.494
	2	0.286	0.365	0.532	0.454	0.409
	3	0.373	0.513	0.557	0.540	0.496
	4	0.270	0.415	0.465	0.439	0.397
	outlet	0.372	0.440	0.605	0.594	0.503
Paint	1	0.401	0.529	0.673	0.640	0.561
Trading Bay	1	0.255	0.575	0.786	0.844	0.615
12 Mile North		–	0.499	0.569	0.621	0.563
12 Mile South		–	0.432	0.566	0.529	0.509
Beech	1	0.353	0.451	0.573	0.592	0.492
Duck	1	–	0.521	0.527	0.522	0.523
Head	1	–	0.478	0.540	0.598	0.539
Haliburton	12	–	0.463	0.628	0.559	0.550
Moose	1	0.348	0.501	0.604	0.621	0.519
	Mean	0.343	0.432	0.543	0.555	0.475
n		31	36	36	36	36

Runoff from 12.8% (Red Chalk) to 56.8% (Blue Chalk) of the watersheds of the study lakes was not measured directly.

Estimates of runoff from these ungauged areas were calculated using the mean runoff of the gauged watersheds for each lake (Blue Chalk and Red Chalk Lakes were treated together).

Annual streamflow volume for each of the 6 lake outlets, 21 gauged inlets and the ungauged areas is summarized in Tables A1-A6, with the relative importance of each to the water balance of the lake. These data are used subsequently to calculate nutrient mass balances for each lake.

iii) ENERGY BALANCE AND EVAPORATION

Evaporation was calculated as the residual term in the energy balance equation of each lake:

$$LE = (R - \Delta T)/(1 + B) \qquad (2)$$

Where: LE = evaporative heat loss by the lake
R = net radiation
ΔT = change in heat storage by the lake
$B = H/LE$, H = sensible heat loss by the lake
B may be estimated independently (Bowen 1926)

Annual lake evaporation rates for the six "A" lakes are summarized in Table 12. Mean annual values were

Table 12. Annual lake evaporation (m yr^{-1}) for 7 study lakes for the 1976-77, 1977-78, 1978-79, and 1979-80 periods.

	Annual Lake Evaporation				
Lake	1976-77	1977-78	1978-79	1979-80	Mean
Blue Chalk	0.759	0.584	0.682	0.601	0.657
Chub	0.808	0.547	0.670	0.666	0.673
Dickie	0.768	0.623	0.731	0.649	0.693
Harp	0.730	0.559	0.676	0.634	0.650
Jerry	0.770	0.570	0.694	0.655	0.672
Red Chalk (main)	0.728	0.566	0.645	0.663	0.651
Red Chalk (east)	–	0.578	0.683	0.657	0.639
Mean	0.761	0.575	0.683	0.646	0.662

0.76 m yr^{-1}, 0.58 m yr^{-1}, 0.68 m yr^{-1} and 0.65 m yr^{-1} in 1976-77, 1977-78, 1978-79 and 1979-80 respectively. The 4-year mean value (0.66 m yr^{-1}) compared favourably with the long-term mean annual value of 0.70 m yr^{-1} for the study area (Hydrological Atlas of Canada 1978).

Differences in annual lake evaporation between lakes were not significant (ANOVA, $p > 0.05$). However, differences between lakes were significant in the months of May, June, July and October, possibly because of the differing heat storage capacities of the lakes. Year-to-year variability in lake evaporation was due to corresponding annual differences in net radiation.

Volumes of water evaporated from the study lakes and the relative importance of this component in the water balance are summarized in Tables A1-A6.

iv) GROUNDWATER

The paucity of surficial material in most of the study watersheds and the impervious nature of the bedrock make it unlikely that groundwater was an important component in the water balance of the study lakes. The intermittent nature of most streams supported this assumption. As other studies of lakes on the Precambrian Shield have demonstrated the relative insignificance of groundwater in the water balance (Schindler et al. 1976), it was assumed that there was no ground water flux to the lakes.

v) STORAGE

Annual changes in lake storage were estimated by the lake level changes, and were generally small, ranging from –0.23 m yr^{-1} to +0.16 m yr^{-1}. Only Harp Lake levels were artificially controlled. Lake levels generally declined in 1976-77, rose in 1977-78 and 1978-79, then declined again in 1979-80. Changes in lake level over four years ranged from –0.06 m to +0.15 m.

Changes in lake storage are shown in Tables A1-A6.

vi) WATER BALANCE

Water balances for the four annual periods and from June 1976 to May 1980 are summarized in Tables A1-A6 for Harp, Jerry, Chub, Dickie, Blue Chalk and Red Chalk Lakes, respectively. Surface runoff supplied the greatest amount of water to all lakes, ranging from 89.3% of the total supply in Jerry Lake to 50.6% in Blue Chalk Lake. Precipitation to the lake surface supplied 10.7% – 49.4% of the total supply. Loss of water to the outlet ranged from 72.1% of the total water loss in Blue Chalk Lake to 92.8% in Jerry Lake, the remainder being lost to evaporation.

Table 13. Percentage correction required to balance lake water budgets.

Lake	Component of Water Balance	Year 1976-77	1977-78	1978-79	1979-80	1976-80
Blue Chalk	Precipitation	+18.9	+46.0	+ 9.0	+ 20.4	+21.5
	Runoff	+18.3	+38.1	+ 10.2	+ 19.6	+21.0
	Evaporation	–20.5	–64.6	– 17.9	– 40.1	–34.2
	Outlet	–12.0	–26.0	– 6.3	– 11.6	–13.3
	Δ Storage	–68.1	–241	–284	–708	–1,430
Chub	Precipitation	–32.1	+33.9	+ 31.5	+ 46.6	+23.7
	Runoff	– 9.2	+ 7.4	+ 10.5	+ 12.4	+ 6.5
	Evaporation	+32.7	–50.7	– 63.4	– 82.3	–36.7
	Outlet	+ 9.9	– 6.6	– 8.4	– 9.8	– 5.6
	Δ Storage	+564	–369	–721	–590	–16,700
Dickie	Precipitation	+19.3	+34.8	– 11.2	– 32.2	– 2.1
	Runoff	+ 9.2	+13.9	– 6.2	– 13.5	– 1.0
	Evaporation	–21.4	–45.0	+ 20.7	+ 58.5	+ 3.2
	Outlet	– 8.1	–11.9	+ 5.2	+ 12.2	+ 0.8
	Δ Storage	N/A[1]	–249	+758	+246	+400
Harp	Precipitation	–30.9	+45.4	+ 43.1	+ 14.7	+19.6
	Runoff	–10.1	+13.0	+ 11.2	+ 3.1	+ 5.1
	Evaporation	+38.3	–73.5	– 72.8	– 21.9	–29.3
	Outlet	+10.6	–10.9	– 9.3	– 2.7	– 4.5
	Δ Storage	N/A[1]	–280	–2,040	– 136	–1,100
Jerry	Precipitation	+60.5	+48.2	+ 82.5	– 89.8	+25.4
	Runoff	+ 8.3	+ 6.3	+ 9.8	– 9.1	+ 3.0
	Evaporation	–71.0	–76.6	–135.3	+129.0	–36.7
	Outlet	– 7.6	– 5.7	– 8.5	+ 9.5	– 2.8
	Δ Storage	–588	–753	–3,140	+595	–870
Red Chalk	Precipitation	+49.6	+45.2	+ 48.3	+ 56.0	+50.1
	Runoff	+10.5	+ 7.9	+ 11.3	+ 11.4	+10.4
	Evaporation	–56.1	–64.8	– 99.8	– 99.9	–80.0
	Outlet	– 9.3	– 7.1	– 9.2	– 9.5	– 8.9
	Δ Storage	N/A[1]	–324	–2,040	–3,130	–1,670

[1] N/A: storage was not measured.

Annual balances were achieved to within ±10% in all 24 cases (6 lakes × 4 years) except for Blue Chalk Lake in 1977-78 (–19.7%). A negative balance indicates that loss from the lake exceeded measured supply to the lake. Over the 4-year period, water budgets balanced to within ±10% in all lakes except Blue Chalk (–10.1%), with 4 of 6 lakes balancing to within 5%. Errors in any of the terms of the water balance could potentially account for the imbalance. Because lake storage, precipitation and evaporation are relatively small components in the balance, large errors in these terms (>100% for storage and >30% for precipitation and evaporation) would be necessary in most cases to explain the imbalance (Table 13). While relatively small errors (<15%) in terrestrial runoff and loss via the outlet could explain the imbalance in most cases, the bias towards negative balances (19 of 24 cases) may indicate unmeasured groundwater flux into the lakes, especially into Blue Chalk and Red Chalk Lakes. However, the relatively good balances obtained give confidence in the individual supply and loss terms and therefore in the use of the data for the construction of chemical mass balances and for testing the mass balance model.

vii) AREAL WATER LOAD (q_s)

The hydrologic data required for testing or using the chemical mass balance model can be expressed in several ways; as an areal water load ($q_s = Q/A_o$), as a lake water replenishment time ($\tau_\omega = V/Q$) or as a water replenishment rate or flushing rate ($\rho = Q/V$), where Q = lake outlet discharge, V = lake volume and A_o = lake surface area. Since these data are interrelated by the lake morphometry, any one can be considered and used in the mass balance model.

The areal water load (q_s) and the lake water replenishment time (τ_w) for each of the study lakes are summarized in Table 14. Mean values of q_s over the 1976-1980 period ranged from 1.69 m yr^{-1} to 8.69 m yr^{-1}. The coefficient of variation between years ranged from 19% to 29% (mean 23%), so that 95% confidence intervals for q_s were equal to the mean value of q_s ± 30% to ± 46% for different lakes. There were highly significant differences in q_s between years ($p < 0.00001$; one-way ANOVA with z-scored data) indicating that more than a single year's data are essential for estimating the hydrologic budget of a lake.

3. MAJOR ION MASS BALANCES

A "conservative" substance is defined with respect to a specific system as one which, at steady-state, has an input that is equal to its output; that is, the system does not act as a source or a sink for the substance. For example, Cl^- is usually considered to be a conservative ion with respect to both terrestrial watersheds (Gjessing et al. 1976) and lakes (Sweers 1969). On the other hand Ca^{2+} is generally exported from watersheds in excess of the input from the atmosphere (Likens et al. 1977, Harvey et al. 1981) but behaves conservatively in soft-water lakes (Schindler et al. 1976). In contrast, phosphorus is retained in both lakes (Vollenweider 1968, Vollenweider and Dillon 1974) and in watersheds (Schindler et al. 1976, Dillon and Rigler 1975).

The elements Ca^{2+}, Mg^{2+}, Na^+, K^+ and Cl^- are generally considered to be conservative with respect to lakes; that is, loss to internal sinks by incorporation into the lake's sediments or biota is insignificant relative to the flux or transport rate through the system. Because of this, their mass balances can be used to check the hydrologic budgets of the lakes, since they, in effect, must act like water in the aquatic system. In other words, it is expected that if the water balance is accurately measured, the inputs and outputs of these 5 ions should be equal for any particular lake.

The mean annual concentration of Ca^{2+}, Mg^{2+}, Na^+, K^+ and Cl^- in each of the study lakes is summarized in Table 15. These results indicate that no major changes in ion concentration have occurred over the course of the study; therefore, the concentrations of these ions are in steady-state. The only exception to this is the Cl^- concentration of Harp Lake; a small increase in concentration has been observed each year, with the result that the concentration in the first year is significantly lower than that measured in the fourth year. However, both the Cl^- and K^+ concentrations in all of the study lakes are extremely low. Since these values, which are close to the analytical detection limits, reflect the concentrations of the major inputs (streams, precipitation), the mass balances of these elements are imprecise.

The major ion mass balances for the six "A" lakes are summarized in Tables A7-A12. The total input including that from all streams, the ungauged area of the watershed and the atmosphere is corrected for change in storage in

Table 14. Areal water loads (q_s, m yr^{-1}) and theoretical water replenishment times (τ_w, years) for "A" study lakes for four 12-month periods, mean annual water loads and water replenishment times and coefficient of variation (c.v., %)

	q_s (m yr^{-1})/τ_w (years)					c.v.
Lake	1976-77	1977-78	1978-79	1979-80	mean	(%)
Blue Chalk	1.30/6.54	1.44/5.90	1.95/4.38	2.09/4.09	1.69/5.23	20
Chub	2.68/3.30	4.22/2.10	5.09/1.74	5.59/1.58	4.40/2.18	29
Dickie	2.04/2.44	2.36/2.11	2.89/1.72	3.12/1.59	2.60/1.97	19
Harp	2.65/4.67	3.78/3.26	5.31/2.33	5.08/2.43	4.20/3.17	29
Jerry	7.17/1.72	7.66/1.61	11.02/1.12	8.90/1.39	8.69/1.46	20
Red Chalk	4.38/3.25	5.17/2.75	7.10/2.00	6.98/2.04	5.91/2.51	23
						23.3 mean

particular. Based on the combined evidence provided by the hydrologic and ion balances, the data collected in the second year is suspect.

The Red Chalk Lake ion and hydrologic balances demonstrated consistent but generally small deficits in input (Tables A6, A12), ranging from 7 to 9% for water and 1 to 18% for the cations.

In almost all cases, particularly for Harp, Chub, Blue Chalk and Red Chalk Lakes, the deficit in the ion budgets is in input. The same trend was measured with respect to the hydrologic budgets, indicating that there is likely a small systematic error in the mass balances of at least these four lakes. There are several possible explanations for these observations. There may be a small input of groundwater (unmeasured) to these lakes. The calculation of the input of water and ions from the ungauged portion of each watershed may be underestimated when calculated by proration of the yields from the gauged watersheds.

In summary, the major ion budgets generally support the contention that the hydrologic budgets were accurately measured. The data collected for Blue Chalk Lake in 1977-78 appear to be the least accurate, and indicate that the TP budget for the same time period is probably poorly measured. The data collected for Chub Lake in 1976-77 and Blue Chalk Lake in 1978-79 and 1979-80 also demonstrated poor ion balances. However, because the hydrologic balances were within 10%, the data collected for these time periods are utilized in the subsequent sections.

4. ATMOSPHERIC DEPOSITION OF NUTRIENTS AND IONS

Atmospheric deposition is a major source of ions to lakes underlain by the sparingly soluble bedrock of the Precambrian Shield. Several investigators have shown that this is particularly true for H^+, NH_4^+, and NO_3^- (reviewed by Scheider et al. 1979). Atmospheric deposition may also be a major pathway for introducing nutrients to softwater lakes. Previous studies of TP budgets for Precambrian Shield Lakes in Ontario, for example, have reported that 5-70% of the TP input to lakes was directly supplied by precipitation to the lake surface (Dillon 1974, Schindler et al. 1976, Cross 1977, Scheider 1978, Scheider et al. 1979).

In this section, the spatial and temporal variability of the atmospheric deposition rates of ions and nutrients, particularly phosphorus, to the study lakes are summarized. Best estimates of annual deposition rates for use in mass balance calculations are generated. Sources of materials deposited from the atmosphere are investigated by examining seasonal trends in deposition and comparing supply rates of materials in "wet-only" (during precipitation events) and "bulk" (wet plus dry) collections.

i) SPATIAL VARIABILITY

Jeffries et al. (1978) found that variations in deposition rates were not significant among five collectors located on a single lake (Red Chalk Lake) for TP, TKN, NO_3^-, H^+, Ca^{2+}, Mg^{2+}, Na^+, K^+, SO_4^{2-} and Cl^-. Significant differences were observed between collectors located on the lake and those located under the forest canopy adjacent to the lake. Forest canopies certainly modified precipitation chemistry, increasing TP concentrations, for example.

Spatial variability in deposition on a regional scale in the study area was assessed by comparing deposition rates of all nutrients and major ions measured at five of the nine collector locations indicated in Figure 2. These comparisons were made only when all collectors were operating.

Deposition rates of precipitation and all materials measured at all but one of the stations, the most easterly one located in Algonquin Park, were not significantly different ($p > 0.05$ in ANOVA analyses). Therefore, data from all but the Algonquin Park station were pooled to provide an average estimate of deposition applicable to the entire Muskoka/Haliburton area.

ii) TEMPORAL VARIABILITY IN BULK DEPOSITION

Bulk deposition is defined as that collected in a continuously open collector. A summary of the range and median values for monthly bulk deposition (mg or meq m^{-2} mo^{-1}) and corresponding volume-weighted concentrations is given in Table 16. The degree of variation in monthly values reflects the combined variation of precipitation depth and the samples' chemical concentrations. Minimum values in precipitation depth were observed in mid-winter and spring and maximum values in autumn and early winter.

The pattern of temporal variation often indicates the source of materials in the bulk deposition. Seasonal patterns were observed for TP (Figure 3), NH_4^+, TKN, Ca^{2+}, Mg^{2+}, Na^+, K^+ and SO_4^{2-}. The deposition of all of these substances was greatest in the warm months of the year. As phosphorus deposited from the atmosphere is

Table 16. Range and median of monthly bulk deposition of ions and nutrients measured at Muskoka-Haliburton from June 1976 to May 1980. Corresponding volume-weighted concentration is also presented. Deposition and concentration are expressed as meq m^{-2} $month^{-1}$ and meq m^{-3} respectively for ions and as mg m^{-2} $month^{-1}$ and mg m^{-3} respectively for nutrients.

	Monthly Bulk Deposition		Volume-Weighted Concentration	
Parameter[1]	Range	Median	Range	Median
H^+	1.00-18.5	5.49	17-136	72
Ca^{2+}	0.37-5.70	1.74	5-102	30
Mg^{2+}	0.12-2.96	0.53	2-35	7
Na^+	0.13-4.31	0.80	1-42	13
K^+	0.02-3.60	0.17	0-43	2
NH_4^+	0.35-7.77	2.29	6-75	30
SO_4^{2-}	0.80-18.0	5.61	19-197	77
Cl^-	0.12-1.32	0.52	3-34	7
NO_3^-	0.81-8.83	2.74	9-89	43
TP	0.35-21.9	1.52	7-472	24
DP	0.26-7.34	0.73	3-140	11
NO_3^--N	11.4-124	38.4	123-1247	605
NH_4^+-N	4.92-109	32.1	77-1050	418
TKN	10.4-166	48.3	172-2190	700

[1] Note that NO_3^- and NH_4^+ have been included both as ions (meq m^{-2} mo^{-1} or meq m^{-3}) and nutrients (mg m^{-2} mo^{-1} or mg m^{-3}).

Table 15. Mean annual concentration of major ions in the study lakes. Sample size varies from ≈20-40; 95% confidence intervals are also shown.

		Ca^{2+}			Mg^{2+}			Na^{+}			K^{+}			Cl^{-}		
Blue Chalk	1976-77	2.82	±	0.10	0.80	±	0.03	0.84	±	0.04	0.38	±	0.02	0.43	±	0.02
	1977-78	2.90	±	0.11	0.79	±	0.02	0.83	±	0.04	0.38	±	0.02	0.47	±	0.02
	1978-79	3.03	±	0.16	0.75	±	0.03	0.83	±	0.05	0.37	±	0.02	0.46	±	0.04
	1979-80	2.88	±	0.10	0.68	±	0.02	0.84	±	0.06	0.39	±	0.02	0.47	±	0.01
Chub	1976-77	2.86	±	0.09	0.75	±	0.02	0.72	±	0.05	0.45	±	0.02	0.56	±	0.03
	1977-78	2.97	±	0.12	0.76	±	0.02	0.79	±	0.04	0.45	±	0.01	0.61	±	0.03
	1978-79	3.07	±	0.12	0.71	±	0.03	0.77	±	0.07	0.42	±	0.01	0.58	±	0.02
	1979-80	2.89	±	0.13	0.64	±	0.02	0.73	±	0.05	0.41	±	0.03	0.60	±	0.06
Dickie	1976-77	2.52	±	0.15	0.64	±	0.04	0.75	±	0.06	0.44	±	0.03	0.79	±	0.06
	1977-78	2.75	±	0.14	0.69	±	0.02	0.85	±	0.05	0.46	±	0.02	0.87	±	0.05
	1978-79	2.82	±	0.10	0.65	±	0.02	0.92	±	0.08	0.43	±	0.01	0.86	±	0.04
	1979-80	2.74	±	0.15	0.58	±	0.03	0.86	±	0.08	0.42	±	0.02	0.87	±	0.07
Harp	1976-77	3.23	±	0.08	1.01	±	0.03	0.94	±	0.05	0.56	±	0.02	0.63	±	0.03
	1977-78	3.21	±	0.10	0.98	±	0.02	0.98	±	0.03	0.56	±	0.02	0.73	±	0.04
	1978-79	3.39	±	0.12	0.96	±	0.04	1.06	±	0.07	0.55	±	0.02	0.76	±	0.03
	1979-80	3.35	±	0.14	0.89	±	0.02	1.03	±	0.06	0.55	±	0.02	0.81	±	0.02
Jerry	1976-77	3.60	±	0.12	1.20	±	0.04	0.86	±	0.07	0.58	±	0.02	0.51	±	0.03
	1977-78	3.55	±	0.09	1.16	±	0.03	0.94	±	0.06	0.56	±	0.02	0.57	±	0.03
	1978-79	3.77	±	0.18	1.17	±	0.06	1.04	±	0.08	0.54	±	0.03	0.54	±	0.04
	1979-80	3.55	±	0.19	1.00	±	0.02	0.90	±	0.08	0.52	±	0.02	0.58	±	0.15
Red Chalk	1976-77	2.80	±	0.08	0.87	±	0.03	0.75	±	0.03	0.44	±	0.01	0.45	±	0.04
	1977-78	2.92	±	0.13	0.85	±	0.02	0.80	±	0.04	0.44	±	0.01	0.46	±	0.02
	1978-79	3.05	±	0.13	0.83	±	0.04	0.82	±	0.07	0.45	±	0.03	0.47	±	0.03
	1979-80	2.92	±	0.14	0.73	±	0.03	0.83	±	0.08	0.43	±	0.01	0.49	±	0.08

the lake as a result of change in lake level. The result is compared to the loss of the ion via outflow from the lake.

Minor components of the mass balance are not considered here, but may alter slightly the results discussed subsequently:

a) accumulation of each ion in the lakes' sediments is an unmeasured loss,
b) input (or output) via groundwater is not considered,
c) input from anthropogenic activities (waste disposal systems, road treatment with salts) is not considered,
d) small (real) changes in lake concentration may result in changes in storage of each ion in the lakes.

The mass balances calculated for the four-year period for Harp Lake (Table A7) show a consistent deficit of 4-8% (excluding K^{+}), comparable to the deficit of 3.6% in the water balance (Table A1). Either a very small systematic error in the hydrologic measurements (overestimation of the outflow volume or underestimation of the inflow volume) has been made or a small unmeasured component of the water balance that contributes ions, e.g. groundwater, may account for the small differences.

The annual mass balances are less precise; variation ranges from about –20 to +20%, either because the accuracy of the measurements is in this range but is averaged out over the longer four-year period, or, more likely, annual changes in water balance are not immediately translated into output because of the finite mixing time and water replenishment time of the lake.

The mass balances obtained from Jerry Lake (Table A8) indicate that the hydrologic balance was accurately measured; Ca^{2+}, Mg^{2+} and K^{+} budgets balanced within 3%. The Na^{+} balance deviated by –7.5%, largely as a result of a very poor balance (–24.5%) in only one year (1978-79), attributable to very high concentrations of Na^{+} measured in the outflow during spring snowmelt that were observed in no other year.

Excellent balances for all four cations (–0.9% to –3.0%) were obtained for Dickie Lake (Table A10). For both Jerry and Dickie Lakes, the Cl^{-} balances were poor, at least in part because the concentrations measured in the inflows and the atmospheric deposition were at or near the analytical detection limits. The balances for all elements were also less precise on an annual basis for the same reasons as for Harp Lake.

The Chub Lake major cation balances (Table A9) were acceptable on a four-year basis (+3 to +10%). However, the balances in the first year were very poor; with an excess in input of 42 to 86%. The second, third and fourth-year balances were good, with, in most cases a slight deficit in input. On the other hand, the hydrologic balance was reasonably good in all years (Table A3). A deficit in input was observed in the last three years (–6 to –9%), but a surplus (+7%) was measured in the first year. The difference in hydrologic balance between the first year and the other three years of ~15% was much less than the differences in the ion mass balances.

The Blue Chalk Lake hydrologic balance was the poorest of the study lakes (Table A5), almost certainly because the gauged portion of the lake's watershed was the lowest proportion of the total watershed of any of the lakes. The deficit in input ranged from 5 to 20%, with the second year being by far the poorest. The ion balances also demonstrated a consistent deficiency in input (Table A11), attributable to the last three individual years in

v) DRY VS. WET DEPOSITION

The relative importance of dry vs. wet deposition is reflected in the wet:bulk deposition ratios. Using only data obtained from simultaneous collections of wet and bulk deposition, median values of wet:bulk deposition are summarized in Table 19 for the nutrients and major ions.

Table 19. Ratio of wet deposition to bulk deposition for ions and nutrients (as %). The reported results are the median values of all collections with equivalent collection periods from the Muskoka-Haliburton network from June 1976 – May 1980. The number of sample pairs is indicated (n).

Parameter	Median Percent	n
H^+	102	269
Ca^{2+}	73	219
Mg^{2+}	70	222
Na^+	100	202
K^+	68	219
SO_4^{2-}	89	229
Cl^-	92	230
TP	47	226
DP	59	160
NO_3^--N	90	192
NH_4^+-N	83	157
TKN	81	195

Wet deposition accounted for 89 – 102% of the bulk deposition for H^+, Na^+, SO_4^{2-}, Cl^- and NO_3^-. In contrast, dry deposition was an important component of the total deposition for Ca^{2+}, Mg^{2+}, K^+, TP, DP, and DOC (wet <75% of bulk). Deposition of the first group of parameters is strongly influenced by the long-range transport of materials from distant sources while the second group is more strongly influenced by the atmospheric suspension of lithologic (soils, dust) and biological (pollen, plant debris) materials from nearby sources.

5. WATERSHEDS AS A SOURCE OF PHOSPHORUS

Phosphorus is transported from terrestrial watersheds to lakes via streamflow. It may be transported in dissolved inorganic or organic forms, as colloidal phosphorus, or in particulate forms ranging from sub-micron size to leaf-litter and other large debris.

The flux of total phosphorus in all forms except the large particulate material has been measured from each of the well- defined subwatersheds of the study ("A") lakes, and from nine additional watersheds (the "export" watersheds) to broaden the range of geological (both bedrock and surficial deposits) types.

Table 20. Areal extent of bedrock and surficial deposit types of the "A" lake subwatersheds and ungauged areas (ung.) expressed as percent. Subwatershed area (A_d) is included. Details in Jeffries and Snyder (1983).

			Bedrock Type (%)				Surficial Deposit Type (%)						
Lake	Sub-water-shed	A_d (ha)	Gneiss (Biotited Horne-blende)	Diorite	Amphi-bolite and Schist	Migma-tite	Minor Till Plain	Thin Till and Rock Ridges	Peat	Bed-rock	Out-wash	Sand	Pond
Blue Chalk	1	27.1	100				94.0	6.0	–	–	–	–	–
	Ung.	100.4	100				76.3	14.9	–	1.8	7.0	–	–
Chub	1	79.0	100				24.2	72.4	2.8	0.6	–	–	–
	2	121.9	100				16.7	75.3	8.0	–	–	–	–
	Ung.	85.0	100				62.2	32.2	0.9	4.7	–	–	–
Dickie	5	34.4				100	–	74.6	25.4	–	–	–	–
	6	25.4				100	–	78.0	22.0	–	–	–	–
	8	71.4				100	13.7	78.1	8.2	–	–	–	–
	10	71.6				100	–	82.8	17.1	–	–	–	–
	11	109.1				100	–	79.1	20.9	–	–	–	–
	Ung.	136.7				100	10.1	86.6	3.2	0.1	–	–	–
Harp	3	22.6	93		7.0		79.5	11.2	9.3	–	–	–	–
	3A	22.4	42.6		57.4		97.1	–	2.9	–	–	–	–
	4	139.0	13.2		86.8		56.0	32.8	–	0.9	–	7.5	2.7
	5	2o4.8	100				34.5	48.6	13.3	–	–	3.6	–
	6	21.5	100				45.2	54.8	–	–	–	–	–
	6A	18.9	33.3	66.7			6.6	84.9	8.5	–	–	–	–
	Ung.	79.4	91.5	1.2	7.3		65.8	27.3	1.4	0.3	–	5.5	–
Jerry	1	7.3	100				77.4	22.6	–	–	–	–	–
	3	666.3	100				13.3	78.4	5.2	0.1	–	2.5	0.4
	4	41.0	100				13.6	83.5	–	–	–	2.8	–
	Ung.	146.0	96.4	3.6			44.7	39.1	–	–	–	8.3	–
Red Chalk	1	157.1	100				53.0	41.1	–	–	0.8	–	4.9
	2	32.9	100				–	67.9	10.5	19.4	–	–	2.2
	3	98.7	100				81.7	2.7	9.9	1.2	–	–	4.5
	4	60.3	100				76.2	16.0	2.9	–	–	–	4.8
	Ung.	85.3	100				48.0	17.1	–	1.2	33.7	–	–

primarily of lithologic (soils, dust) or biological (pollen) origin, minimal deposition rates should be anticipated in the winter when the ground is frozen and snow-covered, and the vegetation inactive or snow-covered.

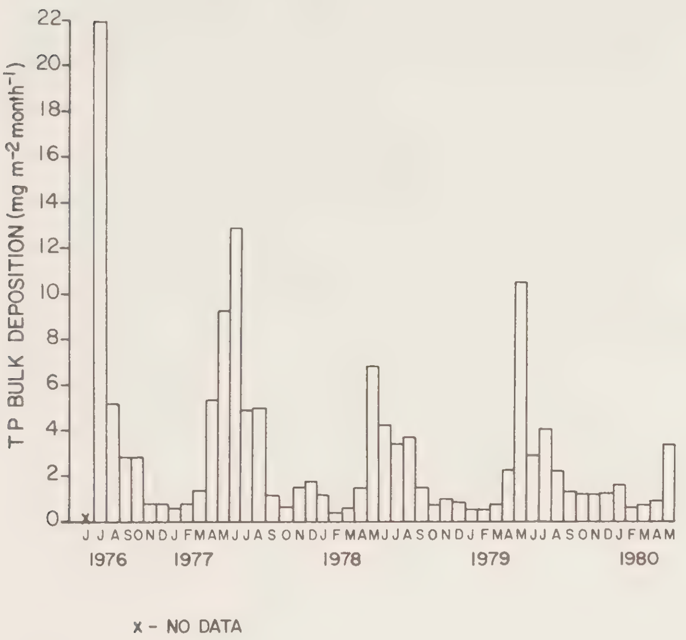


Figure 3 Monthly bulk deposition of total phosphorus (in mg m^{-2} mo^{-1}) measured at Muskoka-Haliburton from June 1976 to May 1980. Note the seasonal pattern (spring or summer maxima) of deposition.

Many of the seasonal patterns in deposition rate simply reflect the pattern of precipitation depth, with concentration of the substance approximately constant. Certain parameters, particularly TP, also showed elevated concentrations in summer thereby increasing the seasonal nature of their deposition. It should be noted that much of the material deposited by the atmosphere in winter months is stored in the snowpack and released quickly during spring melt. In this respect, the period of greatest input to lakes occurs during the spring months.

iii) PRECIPITATION CHEMISTRY

The median concentrations of TP, other nutrients and ions measured in bulk deposition and wet-only precipitation samples are summarized in Table 17. Concentrations ranged over 1 to 3 orders of magnitude for both sample types depending upon the parameter. Sulphate showed the smallest variability while TP, H^+, Na^+, NH_4^+, DP and NO_3^- exhibited large variations in concentration. On an equivalent basis, the order of importance for major ions in both bulk and wet-only samples was $H^+ \simeq SO_4^{2-} > NO_3^- > Ca^{2+} \simeq NH_4^+ > Na^+ > Mg^{2+} \simeq Cl > K^+$. This order of importance is in approximate agreement with that found for other southern Shield locations. Minor differences probably reflect different sampling locations and methodologies.

iv) ANNUAL BULK DEPOSITION

The annual bulk deposition of TP, other nutrients and ions for four annual periods beginning in June1976 is summarized in Table 18. Nearly a three-fold variation in annual deposition of TP was observed, indicating the need for long-term precipitation monitoring to obtain reliable average estimates of deposition. Annual variability in the deposition of other chemical species was as great as four-fold over the 4-year period. However, the variation in annual deposition was small compared to between-month variability.

The TP bulk deposition measured in Muskoka-Haliburton fell within the range of values reported by other studies in North America in general, and those located on Shield terrain in particular. Mean deposition over the 4-year study was 37.3 mg m^{-2} yr^{-1} while the North American median was 33.9 mg m^{-2} yr^{-1}. The median North American deposition rate of NO_3^- was 410 mg m^2 yr^{-1} and of NH_4^+ was 273 mg m^{-2} yr^{-1}. The corresponding 4-year mean values found at Muskoka-Haliburton (537 for NO_3^-, and 408 for NH_4^+) were both higher than these medians.

Table 17. Range and median concentration of ions (μeq L^{-1}) and nutrients (mg m^{-3}) in bulk deposition and wet-only precipitation samples collected at all sites in Muskoka-Haliburton from June 1976 to May 1980.

	Bulk		Wet-Only	
Parameter[1]	Range	Median	Range	Median
H^+	0-457	79	1-1070	76
Ca^{2+}	2-150	29	1-150	24
Mg^{2+}	1-66	6	1-123	4
Na^+	0-174	12	0-113	13
K^+	0-46	2	0-32	1
NH_4^+	0-154	29	2-187	25
SO_4^{2-}	10-323	77	10-252	73
Cl^-	1-138	6	0-113	5
NO_3^-	0-186	36	0-118	31
TP	1-876	20	1-311	10
DP	0-602	8	0-216	4
NO_3^--N	5-2600	500	5-1650	430
NH_4^+-N	5-2150	406	20-2620	356
TKN	35-4960	700	65-3360	575

[1] Note that NO_3^- and NH_4^+ have been included under both ions (as meq m^{-3}) and nutrients (mg-N m^{-3}).

Table 18. Annual bulk deposition (mg m^{-2} yr^{-1}) of ions and nutrients for the composite Muskoka-Haliburton station from June 1976 to May 1980.

	Bulk Deposition (mg m^{-2} yr^{-1})			
Parameter	1976-77	1977-78	1978-79	1979-80
TP	58.4	36.5	30.2	21.8
DP	21.5	21.7	13.9	8.69
NO_3^--N	375	499	602	668
NH_4^+-N	260	369	552	438
TKN	620	598	993	692
H^+	48.0	63.5	101	84.6
Ca^{2+}	552	608	541	341
Mg^{2+}	106	106	70.1	65.0
Na^+	271	444	314	101
K^+	205	136	95.8	83.3
SO_4^{2-}	3,020	3,360	4,040	3,660
Cl^-	263	296	220	211

Table 21. Areal extent of bedrock and surficial deposit types of the "export" watersheds, expressed as %. Watershed area (A_d) is included. Details in Jeffries and Snyder (1983).

		Bedrock Type (%)				Surficial Deposit Type (%)							
Stream Name	A_d (ha)	Biotite Gneiss (Horne-blende)	Marble	Arkose & Minor Marble	Gneiss, Meta Arkose & Marble	Till (Carbo-nate)	Minor Till Plain	Thin Till and Rock Ridges	Peat	Out-wash	Esker	Drumlin	Pond
Twelve Mile North	426.7	–	11.8	–	88.2	93.0	–	–	2.6	1.4	0.1	–	2.9
Twelve Mile South	171.8	–	2.2	–	97.8	81.0	–	–	8.6	6.7	–	–	3.7
Beech Lake Inflow #1	571.6	–	20.6	–	79.4	89.3	–	–	6.9	–	1.5	0.1	2.2
Haliburton Lake Inflow #12	65.6	100	–	–	–	–	–	–	–	100	–	–	–
Duck Lake Inflow #1	47.3	–	11.9	88.1	–	100	–	–	–	–	–	–	–
Moose Lake Inflow #1	437.9	5.9	94.1	–	–	97.2	–	–	–	–	–	–	2.8
Head Lake Inflow #1	48.3	100	–	–	–	–	13.8	64.2	22.0	–	–	–	–
Paint Lake Inflow #1	21.3	100	–	–	–	–	51.7	38.0	–	5.6	–	–	4.7
Trading Bay Inflow #1	7.9	100	–	–	–	–	55.6	35.5	8.1	–	–	–	–

i) SPATIAL VARIABILITY

Dillon and Kirchner (1975a) have shown that watershed geology influences TP export. Geological characteristics of the 30 individual subwatersheds are summarized in Tables 20 and 21. All are small, predominantly forested and 25 of the 30 lie entirely on Precambrian metamorphic bedrock. The other five are underlain in part by sedimentary, carbonate-containing material.

The flux or gross export of total phosphorus, expressed as the yield per unit area of watershed, is summarized in Tables 22 and 23. The mean (1976-80) annual TP export ranged from 2.0 mg m^{-2} yr^{-1} (Blue Chalk 1) to 28.5 mg m^{-2} yr^{-1} (Dickie 6). Considering individual years, the range was greater (1.06 – 45.2 mg m^{-2} yr^{-1}).

Table 22. Summary of gross TP export from 21 subwatersheds of the study lakes.

Watershed		TP Export (mg m^{-2} yr^{-1})				
		1976-77	1977-78	1978-79	1979-80	Mean
Blue Chalk	1	1.59	1.73	1.06	3.64	2.01
Chub	1	1.97	2.84	2.91	2.67	2.60
	2	12.2	12.9	11.7	17.4	13.6
Dickie	5	18.8	14.1	11.2	17.7	15.5
	6	14.9	23.2	30.5	45.2	28.5
	8	4.67	5.51	3.25	6.95	5.10
	10	21.4	15.5	12.3	14.0	15.8
	11	6.46	8.13	6.68	8.52	7.45
Harp	3	14.6	10.9	14.4	12.6	13.1
	3A	2.10	3.95	3.24	3.30	3.15
	4	5.35	7.48	8.56	7.87	7.32
	5	11.7	8.33	11.7	10.4	10.5
	6	3.47	3.78	4.59	4.07	3.98
	6A	3.61	3.88	4.43	9.76	5.42
Jerry	1	3.95	3.91	5.14	5.18	4.55
	3	10.7	7.88	11.2	8.90	9.67
	4	4.69	2.99	4.70	4.68	4.27
Red Chalk	1	3.97	5.93	6.10	7.99	6.00
	2	3.97	5.14	6.10	5.07	5.07
	3	4.37	7.94	6.52	6.28	6.28
	4	8.82	10.6	8.60	8.18	9.05
Mean		7.78	7.93	8.32	10.0	8.52
Standard deviation		5.79	5.20	6.27	9.12	6.17

Table 23. Gross TP export from 9 "export" watersheds.

Watershed	TP Export (mg m^{-2} yr^{-1})				
	1976-77	1977-78	1978-79	1979-80	Mean
Paint #1	2.02	2.33	2.12	2.05	2.13
Trading Bay #1	2.56	3.52	3.84	6.23	4.04
Beech #1	5.86	5.84	6.60	11.2	7.38
Duck #1	–	3.19	2.95	4.43	3.52
12 Mile N	–	6.67	7.61	10.5	8.26
12 Mile S	–	6.34	7.55	8.37	7.42
Haliburton #12	–	6.92	9.47	10.8	9.06
Head #1	–	6.51	10.6	16.6	11.2
Moose #1	5.55	6.94	6.91	7.95	6.83
Mean (metamorphic watersheds)	2.29	4.82	6.51	8.92	6.61
Standard deviation	0.38	2.25	4.16	6.24	4.23
Mean (sedimentary watersheds)	5.70	5.80	6.32	8.49	6.68
Standard deviation	0.22	1.51	1.93	2.65	1.84

Two-way ANOVA was used to evaluate the temporal and between- site variability in TP export. There were no significant differences between years ($p > 0.10$). Differences between individual watersheds and between groups of watersheds (all inlets to each lake forming one group) were significant ($p < 0.01$, $p < 0.001$, respectively). The mean TP export for each watershed group, summarized in Table 24, ranged from 2.0 for Blue Chalk Lake to 14.5 for Dickie Lake, with the other 4 groups averaging 6.2 – 8.1 mg m^{-2} yr^{-1}.

There was no significant difference in TP export between the metamorphic watersheds (21 of the subwatersheds of the study lakes plus 4 of the "export" watersheds) and those which have carbonate-containing sedimentary material in their watersheds (5 "export" watersheds). Bedrock type alone was not related to gross export of TP for the study watersheds, in contrast with the suggestion of Dillon and Kirchner (1975a).

Although year-to-year variability was not significant compared to site-to-site variability, short-term temporal variability was evident. For example, the monthly TP export for each of the Harp Lake subwatersheds (Figure 4) varied by 1-2 orders of magnitude within each subwatershed. The greatest exports occurred during the periods of high streamflow, particularly the spring snowmelt period.

Table 24. TP export from each of the "A" lakes' watershed. Results are the areally weighted averages of all subwatersheds of each lake.

Watershed Group	TP Export (mg m^{-2} yr^{-1}) 1976-77	1977-78	1978-79	1979-80	Group Mean
Blue Chalk	1.59	1.73	1.06	3.64	2.01
Chub	7.09	7.90	7.31	10.0	8.08
Dickie	13.2	13.3	12.8	18.5	14.5
Harp	6.81	6.39	7.82	8.00	7.25
Jerry	6.45	4.93	7.01	6.25	6.16
Red Chalk	5.28	7.40	6.83	6.88	6.60
Year Mean	7.78	7.93	8.32	10.0	

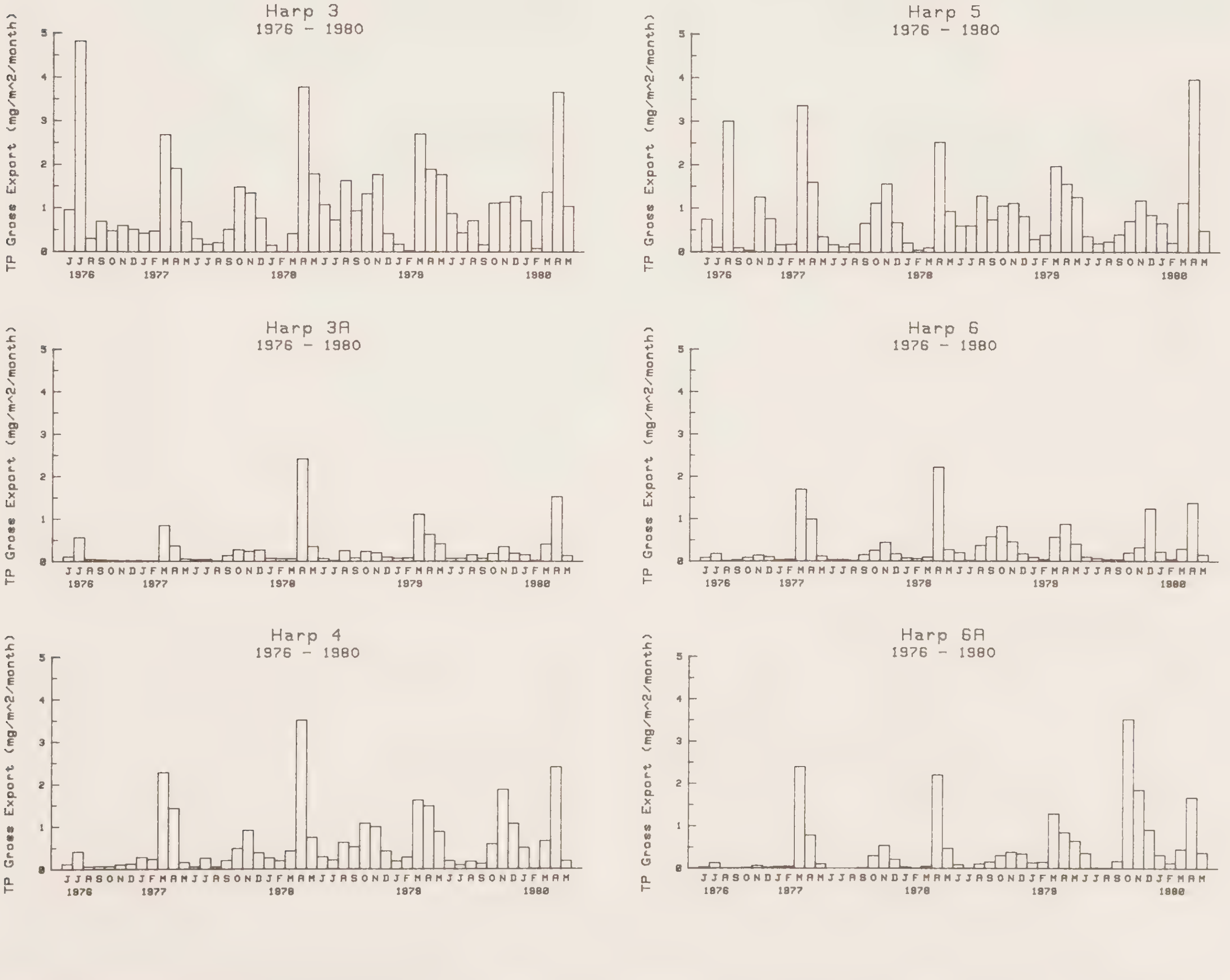


Figure 4 Monthly gross export of total phosphorus (in mg m^{-2} mo^{-1}) measured for six Harp Lake subwatersheds from June 1976 to May 1980.

ii) WATERSHED CHARACTERISTICS AND TP EXPORT

The earliest attempt to classify watersheds on the basis of the total phosphorus export was made by Vollenweider (1968) who noted that watersheds on "unproductive primitive rock" with corresponding "oligotrophic" soils usually had a total phosphorus export of < 20 mg m^{-2} yr^{-1}, while more fertile watersheds, which were often used for agriculture, had TP exports of > 20 mg m^{-2} yr^{-1}. Dillon and Kirchner (1975a) subsequently categorized all watersheds for which TP export was reported according to land use and to bedrock geology. Major land use categories considered were forest, pasture (defined as cleared but uncultivated land), agriculture and urban. Bedrock was separated only into sedimentary and igneous groupings although the latter classification inadvertently included metamorphic watersheds. When only watersheds that were completely forested or had a combination of forest and cleared land with no agriculture were considered, it was found that the TP export could be estimated based on these two land uses and the bedrock geology (Table 25). Watersheds that were forested and on igneous bedrock had the lowest TP export (average of 4.7 mg m^{-2} yr^{-1}; range of 0.7-8.8 mg m^{-2} yr^{-1}), while sedimentary watersheds had higher export if totally forested (mean of 10.7 mg m^{-2} yr^{-1}) or if cleared land was also present (mean of 23.3 mg m^{-2} yr^{-1}).

Table 25. Ranges and mean values for export of total phosphorus from 39 watersheds. Results are in mg m^{-2} yr^{-1} (Dillon and Kirchner 1974).

	Geological Classification	
Land Use	Igneous	Sedimentary
Forest		
Range	0.7-8.8	6.7-14.5[1]
Mean	4.7	10.7[1]
Forest + Pasture[2]		
Range	5.9-16.0	11.1-37.0
Mean	10.2	23.3

[1] Values corrected from original source.
[2] 15% or more of the watershed cleared.

Watersheds whose land use could be classified at least in part as "agriculture" or "urban" had much higher losses of phosphorus, reaching in some cases 1000-2000 mg m^{-2} yr^{-1}. Dillon (unpublished manuscript) subsequently evaluated more recent information using a similar approach. An approximate doubling of the number of watersheds for which total phosphorus export was measured resulted in virtually no change in the TP export-land use-geology relationship (Table 26).

Kirchner (1975) attempted to relate the TP export from one land use-geology class of watersheds (forested, igneous) to physical characteristics of the watersheds. He found that only drainage density (D_d, the total stream length per unit area of watershed, km^{-1}) was significantly correlated to TP export for these watersheds:

$$\text{Export (TP)} = 1.14 + 5.85\, D_d; \; (r^2 = 0.88, n = 18) \qquad (3)$$

Such a relationship can provide the means for more accurate prediction of the TP export for specific watersheds or subwatersheds within the general classification scheme.

Numerous additional watershed studies have been carried out since the work of Dillon and Kirchner. The total phosphorus export from 928 watersheds in the United States was measured as part of the United States National Eutrophication Survey for a 1-year period. Of these, 69 were predominantly ($\geq$90%) forested. The mean TP export from these 69 watersheds was 9.1 mg m^{-2} yr^{-1}, but no distinction was made on the basis of geology (Omernik 1977). Both sedimentary and igneous watersheds were included.

A study of 23 small drainage areas in Finland for a 10-year period (Kauppi 1979) showed that 4.0-5.6 mg TP m^{-2} yr^{-1} were lost from forested watersheds with no human habitation. Many studies (e.g., Jones et al. 1976) have included watersheds where land use was at least in part agricultural or urban. The exports are always higher, often much higher, than in cases where forest predominates. In a number of instances, TP export has been found to be statistically related to anthropogenic activities, e.g., fertilizer usage rates, human or animal population density, etc. (Omernik 1977). However, no new predictive relationships for TP export have been developed for forested watersheds.

Table 26. Ranges and mean values for export of total phosphorus from 69 watersheds. Results are in mg m^{-2} yr^{-1} (Dillon, unpublished manuscript).

	Geological Classification	
Land Use	Igneous	Sedimentary
Forest		
Range	0.7-12.2	6.7-14.5
Mean	5.5	10.3
Forest Pasture[1]		
Range	5.9-16.0	5.8-37.0
Mean	9.8	19.8

[1] 15% or more of the watershed cleared.

The TP export measured for the watersheds of the study lakes and the "export" watersheds have a greater range than previously reported for Precambrian watersheds. However, the mean TP export values for the four years lie within the range summarized in Table 26 in 20 out of 25 watersheds on metamorphic bedrock, with five greater than previously reported, and in four out of five watersheds partially underlain by sedimentary rock, the one measured export (Duck 1) being lower than previously reported. The grand mean TP export of all the metamorphic basins (8.2 mg m^{-2} yr^{-1}) was higher than the mean (5.5 mg m^{-2} yr^{-1}) reported in Table 26 based on previous studies.

6. ANTHROPOGENIC SOURCES OF TOTAL PHOSPHORUS

Human activities may contribute phosphorus (TP) to a lake in many ways. The discharge of TP to a lake from sewage disposal systems, the direct application of TP to

the terrestrial watershed in the form of lawn or garden fertilizers, the alteration of TP supplied to the lake from the watershed because of changes in the land use patterns (e.g., clearing land, constructing roads), the input of TP to the lake by the addition or removal of fill and the mixing of sediment TP into the overlying water through agitation with motorboats are examples of these activities. Human activities must be evaluated on a lake-by-lake basis to estimate anthropogenic TP supply. Phosphorus input from waste disposal systems was considered to be the most important potential source of anthropogenic TP supplied to the study lakes.

Several factors influence the supply of TP to lakes from waste disposal systems; namely, i) the input of TP to the sewage disposal system, ii) the removal or retention of TP by the sewage disposal system, and iii) the transport and retention of TP by the soil and biota between the sewage disposal system and the lake.

i) INPUT OF TP TO SEWAGE DISPOSAL SYSTEMS

The TP supplied to the sewage disposal systems was estimated in three ways. First, a mean value of TP concentration in septic tank effluents was calculated (13.2 mg L^{-1}) based on 21 values in the literature (Table 27). Assuming effluent TP concentration approximates influent concentration, the product of TP concentration and the average per capita water use (164 L/person/day, based on 10 literature values summarized in Table 28) allowed an estimate of 0.79 kg $C^{-1}a^{-1}$ (kilograms per capita per annum). This value agrees closely with a commonly used earlier estimate of 0.80 kg $C^{-1}a^{-1}$ based on 13 studies in Europe and North America (Vollenweider 1968). In the second method, data on TP supply to sewage treatment plants in Ontario (Appleby 1978) and the U.S.A. (Wall and Webber 1970) allowed estimates of 1.4 kg $C^{-1}a^{-1}$ and 1.36 – 1.59 kg $C^{-1}a^{-1}$ respectively. Finally, Bucksteeg (1966) estimated 0.55 kg $C^{-1}a^{-1}$ as the amount of TP ingested in food; presumably a comparable quantity was excreted. The value of 0.55 kg $C^{-1}a^{-1}$ is probably too low as an estimate of TP input to waste disposal systems because it ignores TP in non-food items such as cleaners and detergents. On the other hand, the sewage treatment plant data are probably high because they include TP from human wastes (43% total), industrial and domestic cleaning compounds (29%) and from miscellaneous sources (28%, Appleby 1978).

Table 27. Concentration of total phosphorus ([TP] in mg L^{-1}) in septic tank effluents listed in 21 studies.

[TP]	Source
5	Hansel (1968)
6.5	Preul (1965)
6.9	Ali (1980)
7.7	Barshied and Baroudi (1974)
8.2	Lake George Water Research Centre (1971)
9.8[1]	Ali (1980)
10.0	Feth (1966)
11.6	Viranaghavan and Warnock (1974)
11.8	Brandes (1972)
12.1	Chan (1978)
13.3	Bouma et al. (1972)
14.2	Brandes (1978)
14.6	Otis et al. (1977)
15	Brandes (1977)
15	Brandes (1972)
15.3	Otis et al. (1975)
18[2]	Bennett and Linstedt (1975)
19.6	Brandes (1976)
20	Schroepfer and Preul (1964)
20[3]	University of Wisconsin (1973)
21.8	Harkin et al. (1979)
13.2	mean value (n = 21)

[1] Raw sewage at Toronto's sewage treatment plants.
[2] Average for U.S.A.
[3] Average for Wisconsin.

Table 28. Measured household[1] water use data for year-round dwellings.

Water Usage (L/person/day)	Source
118	Brandes (1976)
151	Otis et al. (1976)
156	Laak (1975)
159	Witt et al. (1974)
161	Siegrist et al. (1976)
168	Chowdry (1974)
173	Chan (1978)
180	Ligman et al. (1974)
184	Harkin et al. (1979)
186	Chan (1978)
164	mean value (n = 10)

[1] Ontario Dept. of Health (1969) estimated 55 L/person/day as usage for cottages. Chan (1978) estimated (no data) 91 L/person/day for cottages. Methods for estimation were not specified.

A value of 0.60 kg $C^{-1}a^{-1}$ can be calculated as the P in human waste from sewage treatment plant data. The value chosen as the TP supply to sewage disposal systems is 0.80 kg $C^{-1}a^{-1}$.

ii) RETENTION OF TP IN SEWAGE DISPOSAL SYSTEMS

The retention of TP by the sewage disposal system varies with the type of system. The four types of systems commonly used in the study area were the septic tank with tile field, the holding tank, the leaching pit and the aerobic system. In the case of a holding tank, all waste is removed from the watershed and, consequently, input of TP to the lake is zero. No TP is removed in the leaching pit or the aerobic system if the compost is eventually emptied within the watershed.

The septic tank and tile field was the most common type of system in use. Sludge accumulation in the septic tank may remove 5% (Brandes 1976) to 20-30% (Hardistry 1974) of the TP in the waste. Further TP removal in the tile field and between the tile field and the lake is dependent upon soil characteristics such as soil texture (proportion sand, silt, clay), soil drainage (frequency of saturation), soil structure (the arrangement of the primary soil particles such as sand and clay into aggregates) and slope (Long 1979) as well as environmental factors such as oxygen, temperature and climatic condi-

tions. Estimates of TP retention in soils in Ontario range from 1% (Brandes 1974) to 99% (Chan 1978). In general, fine-grained soils with high clay content are more effective in the attenuation of TP than are sandy soils or soils with high gravel content. Phosphorus retention coefficients of septic tile fields of differing characteristics are given in Table 29.

The removal of TP by septic systems depends upon their proper maintenance and design. A summary of literature data on the failure rate of septic systems in Ontario (Table 30) showed that an average of 61% were not designed or constructed properly, or maintained in a satisfactory condition.

On the basis of this information, a conservative approach in calculating the potential TP input from anthropogenic sources has been taken and a value of 0.80 kg $C^{-1}a^{-1}$ for septic tank/tile field systems, leaching pits and aerobic systems has been assumed. If it is known that the septic system is in satisfactory condition and information on the soil type is available, then a retention factor may be applied using values outlined in Table 29.

Table 29. The retention coefficients (R_s) of total phosphorus for septic tile filter beds of different characteristics. Results are based on 4 years of data (from Brandes et al. 1974). D_{10} is grain size.

Filter Bed	R_s
1. 22 in. sand (D_{10} = 0.24 mm) 8 in. mixture 4% red mud, 96% sand	0.76
2. 30 in. sand (D_{10} = 0.30 mm)	0.34
3. 30 in. sand (D_{10} = 0.60 mm)	0.22
4. 30 in. sand (D_{10} = 0.24 mm)	0.48
5. 30 in. sand (D_{10} = 1.0 mm)	0.01
6. 30 in. sand (D_{10} = 2.5 mm)	0.04
7. 15 in. sand (D_{10} = 0.24 mm) 15 in. mixture 10% red mud, 90% sand	0.88
8. 15 in. sand (D_{10} = 0.24 mm) 15 in. mixture 50% limestone, 50% sand	0.73
9. 30 in. silty sand	0.63
10. 15 in. sand (D_{10} = 0.24 mm) 15 in. mixture 50% clay-silt, 50% sand	0.74

iii) HUMAN USAGE OF STUDY LAKES

The quantification of human usage of the study lakes was carried out by the Land Use Component, Lakeshore Capacity Study. Results on the number of units, type of sewage disposal facilities and cottage use statistics for the lakes are summarized in Table 31.

No data were available for JerryLake, but very little use is made of the one dwelling on this lake. Use was estimated as 0 capita years yr^{-1}.

Human usage of Red Chalk Lake was calculated as the mean value of usage estimated by three methods, accounting for the observation that two of the three dwellings were mostly vacant during the 1976-1980 period. The 'accessibility equation' (Downing 1983) and the 'cottager equation' (Downing 1983) gave estimates of 0.84 cap yrs yr^{-1} and 0.77 cap yrs yr^{-1} respectively. Extrapolation of usage data for Blue Chalk Lake allowed an estimate of 0.64 cap yrs yr^{-1} as the usage of Red Chalk Lake. The mean value of the three estimates was 0.75 cap yrs yr^{-1}. It was assumed that the dwellings did not utilize holding tanks.

iv) ESTIMATION OF POTENTIAL ANTHROPOGENIC TP INPUTS

The potential anthropogenic TP input for each lake was calculated using the following equation:

$$\text{TP Input (kg yr}^{-1}) = 0.80 \text{ kg TP C}^{-1}\text{a}^{-1} \cdot \text{usage (cap yrs yr}^{-1}) \cdot [1\text{- fraction of dwellings on holding tanks}] \quad (4)$$

Table 30. Summary of cottage pollution survey results obtained by the Ministry of the Environment (from Chan 1978).

Classification[1]	17 Lakes (1971)	29 Lakes (1972)	Shebandowan Lakes (1976)	Muskoka Dickie Lake (1976)	Peterborough District (1975)	Peterborough Victoria (1974)	Muskoka Bass Lake (1974)	Muskoka Bala Lake (1975)
Satisfactory	66	39	37	25	20	61	32	36
Direct Polluter	3	2	2	3	7	–	4	1
Substandard	18[2]	31[2]	40	26	45	39	17	21
Nuisance	13	28	19	39	26	–	38	35
Unclassified	–	–	2	7	2	–	9	7

[1]DEFINITIONS:

Satisfactory — A system which meets all current standards of good design, construction and location, and is properly maintained.

Direct Polluter — A system which permits sewage to contaminate the groundwater, or to reach the lake either by direct discharge through a pipe or ditch or over the ground surface.

Substandard — A system which does not meet current standards of design, construction and location, and/or is in a state of neglect.

Nuisance (Wash Water) — A system causing wash water to be exposed on the surface of the ground either directly through a waste pipe, escaping from a seepage pit or just thrown on ground surface.

Nuisance (Toilet or Solid waste) — A system causing a waste containing fecal or urinary discharges to be exposed on the surface of the ground, either directly through a pipe or escaping from some part of sewage disposal system including a privy.

[2] % of substandard and unclassified systems combined.

Table 31. Results of survey of numbers of occupied dwelling units (No. Units), usage of the units (capita yr yr^{-1}) and sewage disposal types (%) – septic tank and tile field (1), leaching pit (2), holding tank (3) or aerobic system (4). NI indicates information is not available.

Lake	No. Units 1976	1977	1978	Sewage Disposal Type 1	2	3	4	Usage 1976	1977	1978
Basshaunt[4]	8	11	11	83	–	–	17	6.73	9.25	9.25
Bigwind[5]	16	–	18	80	20	–	–	5.98	–	6.72
Blue Chalk[4]	11	–	11	80	20	–	–	7.07	–	7.07
Buck	0	–	0	–	–	–	–	0	–	0
Chub[5]	10	–	11	60	20	20	–	4.01	–	4.41
Crosson[6]	1[3]	1[3]	1[3]	83	–	17	–	–	4.84	–
Dickie[5]	119	–	125	77	10	13	–	108.2	–	113.6
Glen[4]	5	7	7	100	–	–	–	5.59	7.83	7.83
Gullfeather	0	–	0	–	–	–	–	0	–	0
Harp[5]	80	–	83	100	–	–	–	99.8	–	103.5
Jerry	1	–	1	NI	NI	NI	NI	0[1]	–	0[1]
Little Clear	0	–	0	–	–	–	–	0	–	0
Red Chalk	3	–	3	NI	NI	NI	NI	0.75[2]	–	0.75[2]
Solitaire[4]	16	–	14	75	25	–	–	16.7	–	14.6
Walker[5]	23	29	34	100	–	–	–	51.0	64.3	75.4

[1] Estimated, the cottage was very rarely in use.
[2] Estimated, see text.
[3] Ontario Ministry of Natural Resources seasonal camp.
[4] Data on sewage disposal type based on 1976 survey.
[5] Data on sewage disposal type based on 1978 survey.
[6] Data on sewage disposal type based on information from 1977.

A summary of the potential TP inputs for the study lakes is given in Table 32. The lag time between the input of TP to the sewage disposal system and the input of sewage TP to the lake was assumed to be zero. Again, this is a "worst-case" assumption, especially in the case of newly constructed dwellings (e.g., Walker Lake).

Development indices may more appropriately reflect the potential effect of human use on lakes if corrected for lake morphometry. Indices calculated per unit lake area and volume are indicated in Table 33. The ranking of lakes with respect to degree of development does change slightly depending on the index selected. However, of the "A" lakes, Harp and Dickie, have the greatest quantity of development based on all indices, Chub and Blue Chalk an intermediate amount, and Jerry and Red Chalk little or none.

7. MINOR COMPONENTS OF THE TOTAL PHOSPHORUS BUDGETS

In addition to the major components of the phosphorus budgets described individually in the preceding sections, several other inputs and outputs of TP to and from the study lakes may make less significant contributions to the TP budgets.

The harvest of sport fish by anglers represents a loss of TP from the study lakes which may in part be balanced by fish stocking. TP is lost from lakes in the form of emerging insects. It is reintroduced in leaf fall, mainly in the autumn. Nesting birds, if they aggregate in large colonies, may make a substantial contribution to the nutrient budget of lakes.

While none of these pathways of TP was assessed directly in the study lakes, it is possible to estimate the magnitude

Table 32. Potential anthropogenic TP supply to the study lakes from lakeshore waste disposal systems.

Lake	Potential TP Supply (kg TP yr^{-1}) 1976	1977	1978
Basshaunt	5.38	7.40	7.40
Bigwind	4.78	–	5.38
Blue Chalk	5.66	–	5.66
Buck	0	–	0
Chub	2.57	–	2.82
Crosson	–	3.21	–
Dickie	75.3	–	79.1
Glen	4.47	6.26	6.26
Gullfeather	0	–	0
Harp	79.8	–	82.8
Jerry	0	–	0
Little Clear	0	–	0
Red Chalk	0.60	–	0.60
Solitaire	13.4	–	11.7
Walker	40.8	51.4	60.3

Table 33. Indices of development for the study lakes. Use measured in capita-years year^{-1} (cap-yr yr^{-1}).

Lake	Average Usage: Total (cap yr yr^{-1})	Areal (cap-yr ha^{-1}.yr^{-1})	Volumetric (cap-yr 10^4m^{-3}.yr^{-1})
Basshaunt	8.4	0.18	0.023
Bigwind	6.4	0.058	0.005
Blue Chalk	7.1	0.14	0.016
Buck	0	0	0
Chub	4.2	0.13	0.015
Crosson	4.8	0.085	0.010
Dickie	111	1.19	0.238
Glen	7.1	0.44	0.061
Gullfeather	0	0	0
Harp	102	1.52	0.123
Jerry	0	0	0
Little Clear	0	0	0
Red Chalk	0.75	0.013	0.001
Solitaire	15.7	0.13	0.010
Walker	63.6	0.93	0.150

of these additional fluxes. In this section loss of TP in emerging Diptera and in fish harvest corrected for fish stocking is estimated for Harp Lake. Additionally, the significance of leaf litter as a source of TP is assessed for the A lakes.

i) ANGLING AND FISH STOCKING AS TP PATHWAYS FOR HARP LAKE

The quantity of TP removed each year in harvests of sport fish from Harp Lake was estimated as the product of predicted annual harvest (H'', in kg ha^{-1} yr^{-1}), lake area (A_o, in ha) and TP concentration of whole fish. H'' was estimated by solving the sequential series of regression equations developed by McCombie (1982) in the Fisheries Component Lakeshore Capacity Study. Firstly, summer fishing effort by cottagers (E_c) was calculated given A_o = 66.9 ha for Harp Lake, and N (number of dwellings) was 83, as below:

$$\ln E_c = 1.832 + 1.664 \ln A_o - 0.119 (\ln A_o)^2 + 0.2o4 \ln N. \quad (5)$$

E_c was 2o49 angler hrs. E_c/A_o was 30.6 angler hrs ha^{-1}. Summer effort by non-residents (E_n) was calculated from E_c/A_o as below:

$$E_n = 0.518 (E_c/A_o)^{1.2o4} \quad (6)$$

Solving for E_n and adding it to E_c gave a total summer effort $E_s = E_c + E_n$ of 4182 angler hrs or a unit area effort of 62.5 angler hrs ha^{-1}. As there is a winter fishery on the lake, winter effort (E_w) was calculated from

$$E_w = 0.811 (E_s/A_o)^{0.962} \quad (7)$$

and added to E_s to give an annual total effort (E_a) of 7080 angler hr or 106 angler hr ha^{-1} yr^{-1}.

Total harvest (H'') in kg ha^{-1} yr^{-1} was calculated from the morphoedaphic index (MEI = TDS/$\bar{z}$, which was 1.45 g.m^{-4} for Harp Lake), and the calculated E_a:

$$\ln H'' = 0.295 \ln (\text{MEI}) - 0.006 (\ln \text{MEI})^2 + 0.654 \ln(E_a/A_o) - 1.717 \quad (8)$$

The calculated annual fish harvest (wet weight) was 282 kg yr^{-1}.

Harvey (personal communication) indicated that wet to dry ratios of fish from Harp Lake was 4:1. TP concentrations of whole fish average 4% of dry weight (Goodyear and Boyd 1972, Kitchell et al. 1975). Use of these conversion factors indicates that anglers annually remove an estimated 56 kg (dry weight) of fish from Harp Lake containing 2.1 kg of TP. As natural sources alone supply about 70 kg of TP to the lake each year (Table A13, in the Appendix) the influence of angling on TP budgets is small. Angling is probably of less importance in the other study lakes which have less development and poorer winter access.

Two mechanisms related to fish may re-introduce TP that is removed by anglers to the lake. About 30% of E_a is attributable to summer fishing by cottagers (E_c/E_a equals 2o49/7080). Hence about 30% of the removed TP may be redeposited as domestic waste within the Harp Lake watershed. If this fraction is ultimately returned to the lake then the net TP lost from the lake through angling is reduced by 30% to 1.5 kg.

TP lost from the lake by non-resident angling is to some extent balanced by fish stocking. From 1947 to 1981, Harp lake was stocked with speckled, brook and/or lake trout on thirteen occasions. Between 1976 and 1980, when natural TP inputs were measured during this study, 75.9 kg (wet weight) of lake trout were introduced into Harp Lake (Table 34). Using conversion factors identical to those above indicates that an average of 0.14 kg of TP per year was introduced into the lake in this way. Three other study lakes were stocked at somewhat lower intensities (Table 34). These calculations indicate that anglers irreversibly removed an order of magnitude more TP (1.5 kg) than was replaced by planting fish in Harp Lake.

ii) LOSS OF TP BY INSECT EMERGENCE

A second sink for TP in shield lakes is loss of TP in emerging insects. Davies (1980) developed a relationship between J_T/Q (the annual total supply of phosphorus to a lake divided by the outflow volume) and $\bar{B}$ (the lake average dry biomass of emergent Diptera, in g m^{-2} yr^{-1}). The measured annual natural total phosphorus supply to Harp Lake equals about 70 kg, and the average outflow volume equals about 2.8×10^9 L yr^{-1}; hence, J_T/Q averages 25 mg m^{-3} for Harp Lake. Davies reported a relationship between $\bar{B}$ and J_T/Q:

$$\bar{B} = 0.182 + 0.00315(J_T/Q) \quad (9)$$

From this, the average biomass of Diptera emerging from Harp Lake was estimated as 0.261 g m^{-2} yr^{-1}. This yields a total emergence of 174 kg of Diptera ($A_o = 66.9 \times 10^4$ m^2).

Vallentyne (1952) reported an average TP content of adult emergent insects of 0.24% of fresh weight, about 2.4% of dry weight assuming (Davies, pers. comm.) they are 90% water. The 174 kg of emerging Diptera represents a loss of 4.2 kg of P from Harp Lake, a significant quantity in comparison with other sinks such as outflow loss ($\simeq$15 kg) or fish harvest (2.0 kg).

This loss is only temporary, however. Davies (pers. comm.) stated that virtually all emerging adult female Diptera oviposited and subsequently died on the surface of the lake from which they emerged. Males usually also died near or over the water; hence, Dipteran emergence probably represents a very small net TP loss from Harp, or any of the study lakes.

Table 34. Contribution of TP to the "A" lakes from recent (post-1976) stocking of lake trout. Stock records obtained from Fisheries Branch, Ontario Ministry of Natural Resources.

Lake	Number of Fish Stocked	Years of Stocking	Total Weight of Stocked Fish (kg wet wt.)	TP Supplied in Stocked Fish[2] (kg)
Blue Chalk	1500	1976-1978	17.7	0.13
Harp	6100	1976-1980	75.9[1]	0.56
Jerry	6000	1977-1981	51.8	0.38
Red Chalk	5100	1976-1980	55.9	0.41

[1] assuming 2000, 14 month-old fish planted in 1979 had same average individual weight as 15 month-old fish planted in 1980.
[2] calculated assuming 4:1 ratio of wet:dry weight and P content of 3.7% dry weight (see text).

iii) NUTRIENT SUPPLY BY BIRD POPULATIONS

Large nesting colonies of birds may have a substantial impact on lake nutrient dynamics. In fact, the term "guanotrophic" has been coined to describe lakes or ponds whose eutrophic characteristics are attributable to nutrient inputs from birds (Leentvaar 1967, Paloumpis and Starret 1960, Brinkhurst and Walsh 1967). In Ontario, four bird species aggregate in large colonies – herring gulls, ring-billed gulls, Caspian terns and common terns. As none of the "A" lakes had gull or tern nesting colonies, it may safely be assumed that bird faeces was not a significant source of nutrients to these lakes. There was a substantial (>100 birds) nesting colony of ring-billed gulls on Gullfeather Lake, one of the "B" lakes, and clearly an appropriately named lake. The impact of an unmeasured source of nutrients to Gullfeather Lake, attributable to bird faeces, will be discussed in Section V.

iv) NUTRIENT SUPPLY BY FOREST LITTER

Airborne forest litter, especially leaf fall from deciduous forests, may introduce TP to lakes. Hanlon (1981) reported an average input of 260 g dry weight of litter m^{-1} of wooded lake shoreline for four lakes in various parts of the world. Inputs ranged from a low of 135 g m^{-1} for Welsh Llyn Frengoch to a high of 354 g m^{-1} for Mirror Lake in New Hampshire. Assuming forest was continuous in the "A" lake watersheds, leaf-litter inputs were estimated using the mean input given by Hanlon. Estimates ranged from 1010 to 2020 kg of airborne vegetation litter per lake per year (Table 35). From an average TP concentration of leaves of 0.16% dry weight, calculated from Ovington (1956), leaf litter may constitute an estimated 1.6 to 3.2 kg of TP to the study lakes annually. The source is very small in comparison with total measured inputs, e.g., 3% in Harp Lake (Table 35). Leaf litter may become an important source of TP for lakes with very small watersheds, but it may be ignored for the "A" lakes.

Table 35. Estimated total input of airborne vegetation litter to the "A" study lakes. Estimate calculated from shoreline length (L) assuming input is 260 kg km^{-1} of dry material (Hanlon 1981) and TP content of leaves is 0.16% of dry weight (Ovington 1956). Ratio of litter TP supply to natural TP supply is included.

Lake	L (km)	Total Airborne Litter Input: dry matter (kg yr^{-1})	TP (kg yr^{-1})	Ratio
Blue Chalk	4.6	1190	1.9	0.08
Chub	3.9	1010	1.6	0.04
Dickie	7.8	2020	3.2	0.04
Harp	4.6	1190	1.9	0.03
Jerry	4.6	1190	1.9	0.02
Red Chalk	4.8	1240	2.0	0.04

8. LOSS OF PHOSPHORUS FROM THE STUDY LAKES – SEDIMENTATION

Phosphorus is lost from the water column of lakes by two major processes, outflow and sedimentation. The loss of TP from the "A" lakes via outflow was measured accurately by quantification of the discharge (outflow volume per unit time section III.B.2) and the nutrient content of the outflow. The flux of TP to the sediments of lakes is much more difficult to measure; unfortunately, the sediments are usually a more important sink for TP than is loss by outflow. Because of its importance, three independent measurements of TP accumulation in the study lakes' sediments were made.

i) RETENTION OF TP – ESTIMATES USING THE TP MASS BALANCES

The retention coefficient of a lake for TP (Rp) is defined as the fraction of the total input of TP to the lake that is not lost by outflow. If the concentration of TP in the lake is at steady-state (i.e. the concentration is not changing other than on a short-term or seasonal basis), then the retention coefficient is equivalent to the portion of the input lost to the lake's sediments:

$$R_p = \frac{J_T - J_o}{J_T} \qquad (10)$$

where J_T = total input,
J_o = loss by outflow.

For the undeveloped lakes (Jerry and Red Chalk Lakes), the total input of TP (other than that contributed by minor sources – Section III.B.7) was measured directly. The TP concentrations in the lakes were at steady-state (Section III.B.10). Therefore, the retention, or sedimentation rate of TP can be explicitly calculated for these lakes by dividing the mass of TP retained by the sediment area. These calculations are outlined in Table 36.

In Jerry Lake, between 42 and 73% (100 R_p) of the TP input (average of 60%) was lost to the sediments in the individual four years of study, while in Red Chalk Lake, the retention ranged from 47 to 79%, averaging 61%. The retention coefficients and derived sedimentation rates were highly variable from year-to-year, indicating that long-term measurements of mass budgets are essential for accurate calculation of average sedimentation rates.

The TP retention coefficients and sedimentation rates could not be measured directly for the developed lakes because their anthropogenic, and therefore their total input of TP was not known. However, empirical models

Table 36. Sedimentation of TP in undeveloped lakes calculated from the mass balances. Total input (J_T), loss by outflow (J_o), amount retained, sedimentation rate (S), and retention coefficient (R_p) are reported.

Lake	J_T (kg yr^{-1})	J_o (kg yr^{-1})	Retained (kg yr^{-1})	S (mg m^{-2} yr^{-1})	R_p
Jerry					
1976-77	112.3	29.9	82.4	164	0.734
1977-78	79.5	27.7	51.8	103	0.652
1978-79	102.3	45.9	56.4	113	0.551
1979-80	81.7	47.4	34.3	68	0.420
Average	94.0	37.7	56.3	112	0.598
Red Chalk					
1976-77	57.8	12.2	45.6	80	0.789
1977-78	56.2	20.8	35.4	62	0.630
1978-79	50.8	24.6	26.2	46	0.516
1979-80	49.6	26.3	23.3	41	0.470
Average	53.6	21.0	32.6	57	0.608

relating R_p to independent variables (often hydrologic parameters) have been developed and validated for other Precambrian lakes (Kirchner and Dillon 1975, Dillon and Kirchner 1975b, Nurnberg 1983). The information collected for the two undeveloped lakes was therefore used to validate existing models which may then, in turn, be used to predict R_p and S for the developed lakes. This approach is discussed in Section III.B.10.

ii) ESTIMATES USING SEDIMENT TRAPS

In theory, it should be possible to measure the sedimentation rate of TP (or any other substance) by collection of sedimenting particles in containers (sediment traps) positioned above the sediment-water interface. Sediment traps were used to estimate the whole-lake sedimentation of TP in Blue Chalk Lake. Traps were located at eleven sites on the lake over a period of four hydrologic years. Sedimentation rate was calculated at each station for each year. The average of all stations was used as a whole-lake sedimentation rate estimate (T).

Rates of sedimentation of TP are given in Table 37. The TP sedimentation rates calculated in this way ranged from 442 to 640 mg m^{-2} yr^{-1}, with a mean value of 5o4 mg m^{-2} yr^{-1}. These values of T are much greater than those rates (S) calculated from the TP mass balance of the lake. Since the anthropogenic TP inputs were unknown, a range in S was calculated each year assuming that 0 and 100% of the estimated potential anthropogenic input of TP reached the lake. Since this potential input was relatively small (between 17 and 31% of the "natural", measured input), the range in S was not great, averaging 39 to 51 mg m^{-2} yr^{-1}. Rates of sedimentation of TP, measured using sediment traps, were therefore 12 to 17 times greater than that calculated from the measured mass balances. Since T was an order of magnitude greater than the total possible input to the lake (assuming 100% of the estimated potential anthropogenic TP enters the lake), sediment traps clearly do not measure the sedimentation of TP properly.

The usefulness of the sediment trap methodology has been intensely debated (e.g. Hargrave and Burns 1979, Gardner 1980a, 1980b). Similar findings to this study have been reported by Fuhs (1975), who measured a 10-fold difference between TP accumulation in traps and accumulation calculated from the mass balance for Canadarago Lake, New York.

Table 37. Comparison of TP accumulation (mg m^{-2} yr^{-1}) in the sediments of Blue Chalk Lake measured using sediment traps (T), and calculated from the mass balance (S). Standard errors in brackets.

Year	Sedimentation		T/S
	T	S[2]	
1976-77	–[1]	61.4 – 72.8	–
1977-78	640	40.4 – 51.8	12 – 16
1978-79	475	31.0 – 42.5	11 – 15
1979-80	457	23.3 – 34.7	13 – 20
1980-81	442	–[1]	–
Average	504	39.0 – 50.5	12 – 17
	(46)	(8.2) (8.2)	

[1] no data

[2] low end of range calculated assuming input from anthropogenic sources to the lake was 0; the high end of the range calculated assuming all of the potential input from anthropogenic sources reached the lake.

Several factors may contribute to the overestimation of sedimentation of TP by sediment traps. It has been suggested that collection of particulate material in sediment traps interrupts the normal mineralization-production cycling of elements. Other possible phenomena which would result in over-collection of material include the "snow-fence" effect, whereby the horizontal velocity of particles passing the trap-mouth is reduced, resulting in a greater catch. Resuspension of particles already in the surficial sediments is also a common occurrence, especially at spring and fall overturn, and may lead to over-collection of particles in the traps.

The results of this study suggest that sediment traps do not give a quantitative measure of sedimentation rate of TP. Sedimentation rates calculated from sediment traps are therefore not used in the mass balance models.

iii) ESTIMATES USING SEDIMENT CHRONOLOGIES

The sedimentation rate of TP in the lakes was estimated in a third way by using the historical (or paleolimnological) record contained in lake sediments. Recent development of relatively simple sediment dating techniques allows determination of the age of sediment layers deposited in the last few hundred years. Thus, the accumulation rate of matter in the lake's sediments may be calculated. If the nutrient content of these sediments is known, the accumulation rate of the nutrient can therefore be calculated.

To establish the chronologies and thus, the sediment accumulation rates, of lake sediments over the last approximately 100 years, Pb-210 has been shown to be very useful. The rationale for its use has been reviewed by Koide et al. (1973), Robbins (1978) and Krishnaswami and Lal (1978).

Lead-210 is an intermediate product of the U-238 decay series. It is introduced into lakes' sediments by two processes. The decay of Ra-226 in the earth's crust results in the exhalation of Rn-222 to the atmosphere. This isotope rapidly decays to Pb-210, which is, in turn, rapidly removed by wet precipitation and dry fallout, the mean residence time in the atmosphere being only 5-10 days. The residence time of Pb-210 in lakewater is also short because of its affinity for particulates. For the same reason, input of Pb-210 to a lake from its watershed is usually negligible unless transport of particulates (erosion) is an important process. This is very unlikely to be important in Precambrian areas.

The second process resulting in the accumulation of Pb-210 in lake sediments is the decay of the parent Ra-226 in the lakewater and sediments themselves. Since the Pb-210 produced in this way is in secular equilibrium with the Ra-226 in the sediments and water column, this additional activity, which is termed the background or supported Pb-210 activity, will be constant.

Since the flux of Pb-210 to the lake's sediments from the atmosphere is constant and the Pb-210 is not mobile in the sediments, then the change of Pb-210 concentration in the sediments is a function of only radioactive decay and/or change in accumulation rate of lake sediments. Therefore, if no change in sedimentation rate has occurred and there are no processes occurring that result in mixing of the surficial sediments, then the activity of Pb-210 will be log-linear with depth (i.e. with time) until the supported Pb-210 level is reached. Calculation of the sediment accumulation rate (SAR) from the Pb-210 profile is done using the relationship:

$$A(w) = A(m)e^{-\tau w} + A(s) \tag{11}$$

where: A(w) is the Pb-210 activity at any depth defined by w where the mixing zone w depth of supported Pb-210 level
w is the depth scale measured as cumulative dry weight per unit area
A(m) is the activity of Pb-210 at the bottom of the mixing zone (or at the sediment surface if there is no mixing zone)
A(s) is the activity of supported Pb-210
τ is the Pb-210 decay constant ($\tau = \ln 2/t_{1/2} = 0.0311\ yr^{-1}$)

The slope, k, of the regression of ln[A(w)-A(s)] on w is related to the accumulation rate of sediments (SAR in dry weight area^{-1} time^{-1}) by the decay constant:

$$k = \tau t/w \tag{12}$$

where SAR = w/t

$$\text{Therefore, SAR} = \tau/k \tag{13}$$

It must be emphasized that sediment accumulation rates measurements are site-specific; the rate measured at one location cannot be extrapolated to the entire lake or to another site within the lake. However, a recent methodology for establishing whole-lake sediment and nutrient accumulation rates has been proposed by Evans (1980), further developed by Dillon and Evans (1982), and utilized by Evans and Dillon (1982) and Evans et al. (1983).

Dillon and Evans (1982) demonstrated that atmospheric deposition was the only significant source of stable Pb to Precambrian lakes in southern Ontario, and that the Pb was almost entirely of anthropogenic origin (combustion of fossil fuels). Lead deposited on the terrestrial portions of the watersheds was not mobile and did not contribute to that found in the lake. Furthermore, the lakes were extremely efficient sinks or traps for the Pb deposited on their surface; all Pb input to the lakes was retained in the lakes' sediments. The result of these facts is that all the lakes' sediments contain the same mass or "burden" of Pb (per unit area). Whole-lake Pb burdens for lakes in the study area including all of the "A" lakes are shown in Table 38. The mean burden (680 mg Pb m^{-2}) is an estimate of the average amount deposited over the entire study area. Since precipitation from the atmosphere occurs uniformly over the lakes' surfaces over the long-term, there should be no spatial variability in total Pb burden in the lakes' sediments if there was uniform sedimentation of particulate matter. However, sedimentation is not uniform, because of redistribution of sediments to the deeper parts of each lake. The anthropogenic Pb must also be redistributed in a similar, perhaps identical, pattern. Therefore, a comparison of the sediment Pb burden at a particular site to the whole-lake Pb burden must reflect the differential movement of sedimenting particles, and hence the distribution of sediments.

This ratio thus may be employed as a correction factor to estimate whole-lake accumulation from a site-specific measurement of the sediment accumulation rate. Combined with the TP concentration of the sediments at that site the whole-lake TP accumulation rate may then be calculated.

A detailed calculation is shown in Table 39. The sediment accumulation rate measured at four hypolimnetic sites in Jerry Lake was corrected using the whole lake site-specific Pb burden ratio. The resulting average SAR (69 g m^{-2} yr^{-1}) can be combined with the TP concentration of the sediments at the lake's mean depth (1.43 mg g^{-1}, from the relation [TP] = 0.080 (depth) + 0.435) to give an estimated TP accumulation for Jerry Lake of

Table 38. Anthropogenic whole-lake Pb burdens for ten lakes in the study area, including all of the "A" lakes. Standard error of the mean in brackets. "n" is the number of cores from which the whole-lake Pb burden was calculated.

Lake	Pb burden (mg m^{-2})	n
Blue Chalk	541	67
Chub	645	43
Clear[1]	624	48
Crosson[1]	627	34
Dickie	608	39
Harp	814	107
Heney[1]	732	48
Jerry	733	44
Plastic[1]	768	36
Red Chalk	676	92
Mean	677 (26)	

[1] additional lakes studied as part of the Acid Precipitation in Ontario Study.

Table 39. Sediment accumulation rate (SAR(s)) measured at 5 sites in Jerry Lake using the Pb-210 method, Pb burden at the same sites, ratio (F) of whole-lake Pb burden to site-specific Pb burden, corrected sediment accumulation rate (SAR(w)), and TP concentration in the upper 3 cm of sediment at each site.

Site Number	Station depth (m)	SAR(s) (g m^{-2} yr^{-1})	Pb burden (g m^{-2})	F[2]	SAR(w) (g m^{-2} yr^{-1})	[TP] (mg g^{-1})
1[1]	4.8	465	637	1.151	535	0.99
2	11.0	91	965	0.760	64	1.33
3	17.3	118	1262	0.581	89	1.77
4	20.2	109	2139	0.343	63	1.67
5	32.3	173	1034	0.709	59	3.23
Average					69 (14)[3]	

[1] littoral zone core – coarse sediments with very low organic content, excluded from subsequent calculations.
[2] whole-lake Pb burden of 733 mg m^{-2} (Table 38).
[3] standard deviation.

99 mg m^{-2} yr^{-1}. Because the Pb-210 methodology gives an average sediment accumulation rate over a two or three-decade period, this calculation of TP accumulation is, in fact, an estimate of the long-term average.

This result compares favourably with the average value measured for Jerry Lake using the mass balance data (112 mg m^{-2} yr^{-1}; Table 36).

The TP accumulation rates of the "A" lakes calculated using this methodology are reported in Table 40. The results calculated in this manner can only be accurately compared to the results calculated from the mass balances for the undeveloped lakes (Red Chalk and Jerry Lakes). The comparison is less favourable for Red Chalk Lake; 85 mg m^{-2} yr^{-1} vs. 57 mg m^{-2} yr^{-1}.

In summary, independent estimates of TP accumulation in lakes' sediments can be made for the undeveloped lakes using the TP mass balances and for all lakes using the whole-lake sediment accumulation rates and sediment TP concentrations. Sediment traps do not provide meaningful measures of TP flux to lakes' sediments.

Table 40. TP accumulation rates for the "A" lakes calculated from the Pb burden-corrected sediment accumulation rates and the TP concentration of the sediments.

Lake	TP accumulation rate (mg m^{-2} yr^{-1})
Blue Chalk	37
Chub	81
Dickie	74
Harp	85
Jerry	99[1]
Red Chalk	85[2]

[1] compared to 112 mg m^{-2} yr^{-1} (Table 36) calculated from the mass balance.

[2] compared to 57 mg m^{-2} yr^{-1} (Table 36) calculated from the mass balance.

9. TOTAL PHOSPHORUS MASS BALANCES

As discussed in previous sections, the major components of the TP balances of a lake are the flux from the terrestrial watershed either in streamflow or as ephemeral runoff draining the "ungauged" areas of the watershed, the flux from the atmosphere to the lakes, the loss of TP from the lakes by outflow or by sedimentation to the lake bottom, and possibly the input of TP derived from anthropogenic sources (mainly septic systems) into the developed lakes. In this section, the TP mass balances of the "A" lakes are discussed in order of decreasing potential contribution of anthropogenic TP sources. The mass balances are summarized in Tables A13-A18 and presented in Figures 5 to 10.

i) HARP LAKE

Over the four-year survey period, the six gauged inflows to Harp Lake contributed an average of 55% of the measured (i.e., the natural) TP input to the lake, with individual streams providing from 1% (inflow 3A) to 32% (inflow 5) of the total. The amount of TP contributed by any stream varied substantially from year to year, with differences of greater than 100% measured in one case (inflow 6A). The relative importance of the different streams was very consistent from year to year, in part because the larger watersheds provided more runoff and therefore more TP. However, small watersheds may be important contributors to the TP content of lakes. For example, two inflows contributed more TP to Harp Lake than inflow 3, but none had as high a volume-weighted TP concentration (Table 41). Therefore, inflow 3 provided the most TP per volume of water displaced in Harp Lake of all inflows. On the other hand, inflow 3A provided the least amount of TP to the lake, and the least per unit volume of water displaced.

The ungauged area of the Harp Lake watershed contributed an estimated 8.5% (range 6.7-10.7%) of the total input. This amount is relatively small; that is, >90% of the "measurable" inputs were measured, providing confidence in the accuracy of the measurement of the TP inputs.

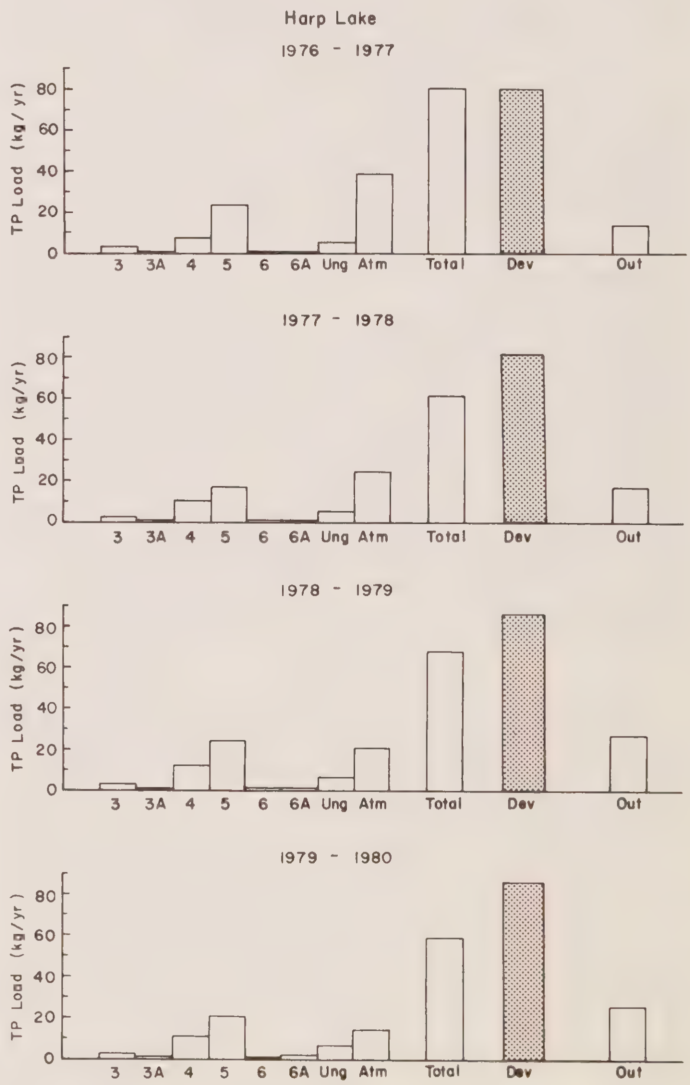


Figure 5 Total phosphorus loading to Harp Lake, in kg/year for four hydrologic years (1976-1980). Inputs are measured streamflow (stream identification number), ungauged area (Ung), atmospheric loading to the lake surface (Atm) and calculated supply from waste disposal systems (Dev). The sum of the natural inputs (Total) and loss by outflow (Out) are also given.

Atmospheric deposition was a very significant source of TP, providing ≃36% of the measured input (range 25-48%) of TP during the study period. The volume-weighted concentration was much greater than that of any of the inflows (Table 41); therefore, atmospheric deposition provided the greatest net input of TP (amount of TP supplied per unit of lake volume displaced) of all natural sources.

The potential supply of TP from anthropogenic sources (section III.B.6) was greater than the sum of all natural inputs (326 kg TP in four years vs. 270 kg from natural sources). If all of the potential anthropogenic TP input actually reached Harp Lake, it would represent 55% of the total input, by far the most of any single source.

The loss of TP by outflow from Harp Lake was relatively small compared to even the natural inputs. Because the TP concentration in the lake was at steady-state during the study period, these results provide further evidence that loss by sedimentation is the major removal mechanism for TP from the lake.

Table 41. Input of TP to Harp Lake per unit volume of water from the natural sources (J/Q) and total input from each source (J). Results are mean values for 1976-80 period.

Source	J/Q J (mg m^{-3})	J (kg yr^{-1})
Inflow 3	23.5	3.0
Inflow 3A	6.3	0.7
Inflow 4	15.0	10.2
Inflow 5	20.2	21.5
Inflow 6	13.7	0.9
Inflow 6A	14.4	1.0
Ungauged	15.9	5.8
Atmosphere	37.7	24.6

ii) DICKIE LAKE

The TP mass balance of Dickie was similar to that of Harp Lake in several respects. The five gauged tributaries contributed an average of 40% of the natural input of TP, while atmospheric deposition contributed 38%. The ungauged area provided a greater proportion of the TP input than in the case of Harp lake (22% vs. 8.5%).

The relative importance of the different tributaries was again different if the volume-weighted concentration was considered instead of the absolute input (Table 42).

Table 42. Input of TP to Dickie Lake per unit volume of water from the natural sources (J/Q) and total input from each source (J). Results are mean values for 1976-80 period.

Source	J/Q J (mg m^{-3})	J (kg yr^{-1})
Inflow 5	28.3	5.3
Inflow 6	56.3	7.2
Inflow 8	14.1	3.7
Inflow 10	27.2	11.3
Inflow 11	17.6	8.1
Ungauged	30.0	7.5
Atmosphere	35.1	8.8

Inflow 6 contributed most to the TP balance according to the former methodology followed by inflows 5 and 10, while inflow 10 followed by inflow 11 contributed most TP in absolute terms.

The potential input of TP from anthropogenic sources was 87% of the natural TP input; that is, if the anthropogenic sources contributed their full potential input, they would constitute 46% of the total input.

Outflow loss of TP was small compared to natural input, again demonstrating the importance of sedimentation.

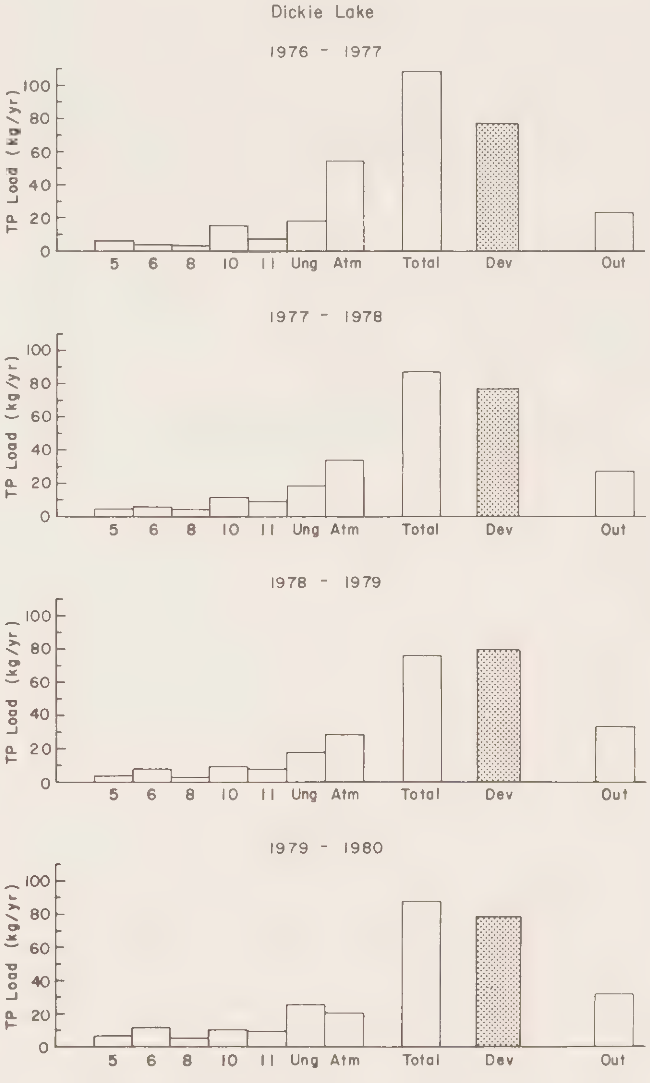


Figure 6 Total phosphorus loading to Dickie Lake, in kg/year for four hydrologic years (1976-1980). Inputs are measured streamflow (stream identification number), ungauged area (Ung), atmospheric loading to the lake surface (Atm) and calculated supply from waste disposal systems (Dev). The sum of the natural inputs (Total) and loss by outflow (Out) are also given.

iii) CHUB AND BLUE CHALK LAKES

The two gauged inflows of Chub lake contributed 50% of the natural TP input over the four-year period; atmospheric deposition contributed an additional 32%. The unmeasured portion of the TP input (18%) was comparable in importance to that of Dickie Lake.

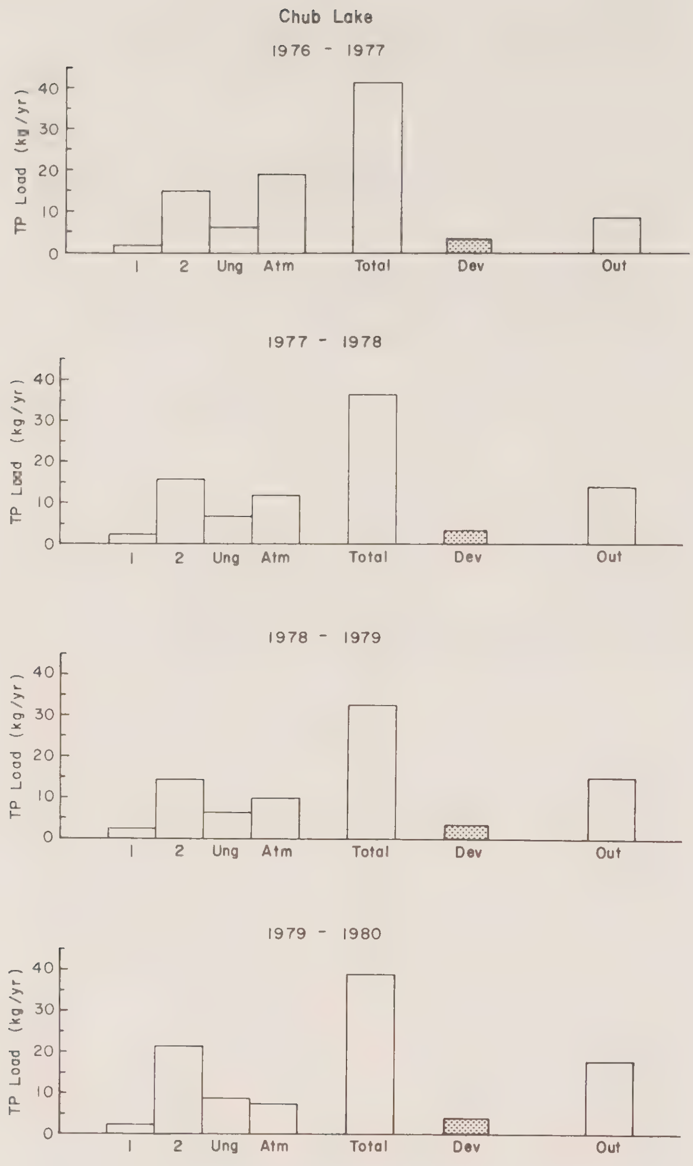


Figure 7 Total phosphorus loading to Chub Lake, in kg/year for four hydrologic years (1976-1980). Inputs are measured streamflow (stream identification number), ungauged area (Ung), atmospheric loading to the lake surface (Atm) and calculated supply from waste disposal systems (Dev). The sum of the natural inputs (Total) and loss by outflow (Out) are also given.

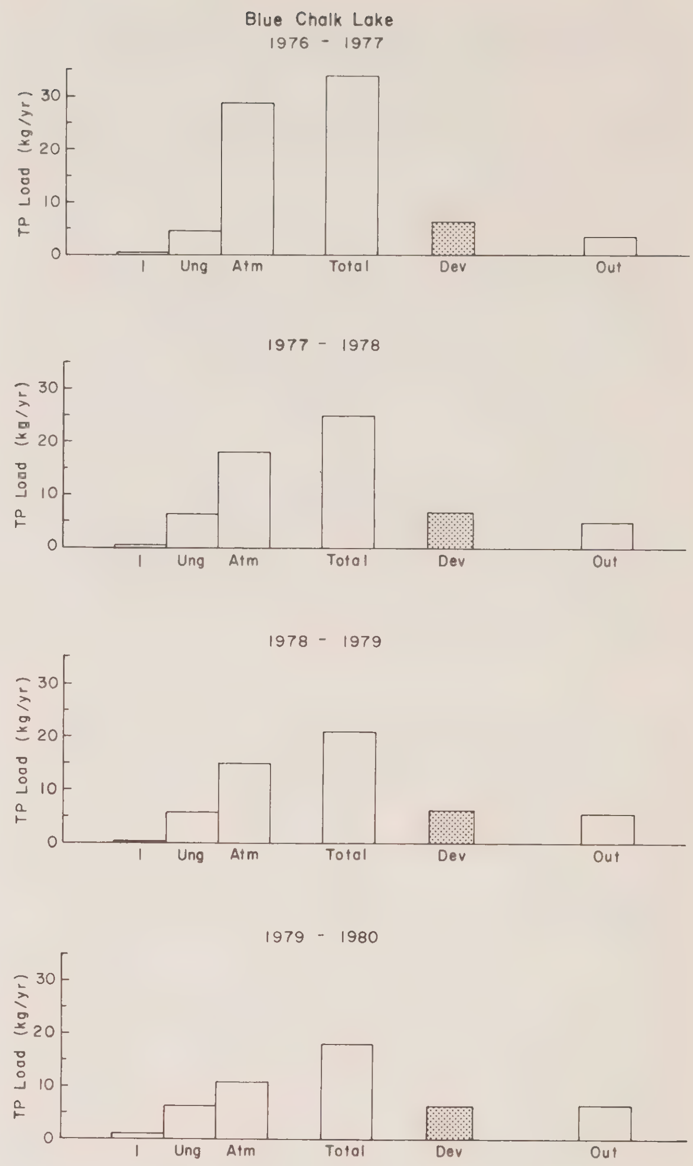


Figure 8 Total phosphorus loading to Blue Chalk Lake, in kg/year for four hydrologic years (1976-1980). Inputs are measured streamflow (stream identification number), ungauged area (Ung), atmospheric loading to the lake surface (Atm) and calculated supply from waste disposal systems (Dev). The sum of the natural inputs (Total) and loss by outflow (Out) are also given.

The potential input of TP from anthropogenic sources was much less important than in the cases of Harp and Dickie Lakes, averaging 7.3% of the measured natural input, or 6.8% of the total potential input of TP.

Loss of TP by outflow was about 37% of the natural input, with sedimentation again necessarily the more significant loss mechanism.

The one gauged tributary of Blue Chalk Lake provided only 2.2% of the natural input of TP. Atmospheric deposition provided by far the major portion of the TP input (≃75%), while the ungauged area contributed an estimated 23%. Anthropogenic sources could potentially provide 23% as much TP as the combined natural sources (19% of the potential total input). About 20% of the natural input was lost by outflow.

iv) RED CHALK AND JERRY LAKES

The four gauged tributaries of Red Chalk Lake contributed 42% of the natural input of TP; the outflow of Blue Chalk Lake contributed an additional 10%. Atmospheric deposition supplied 39% of the TP input, with the ungauged area supplying 9%. The potential contribution from the lakeshore development was insignificant.

The TP mass balance of Jerry Lake was similar to that of Red Chalk Lake; gauged tributaries were the most important source of TP (71%) followed by atmospheric deposition (20%), then by runoff from the ungauged portion of the watershed (10%). There was no potential input from anthropogenic sources.

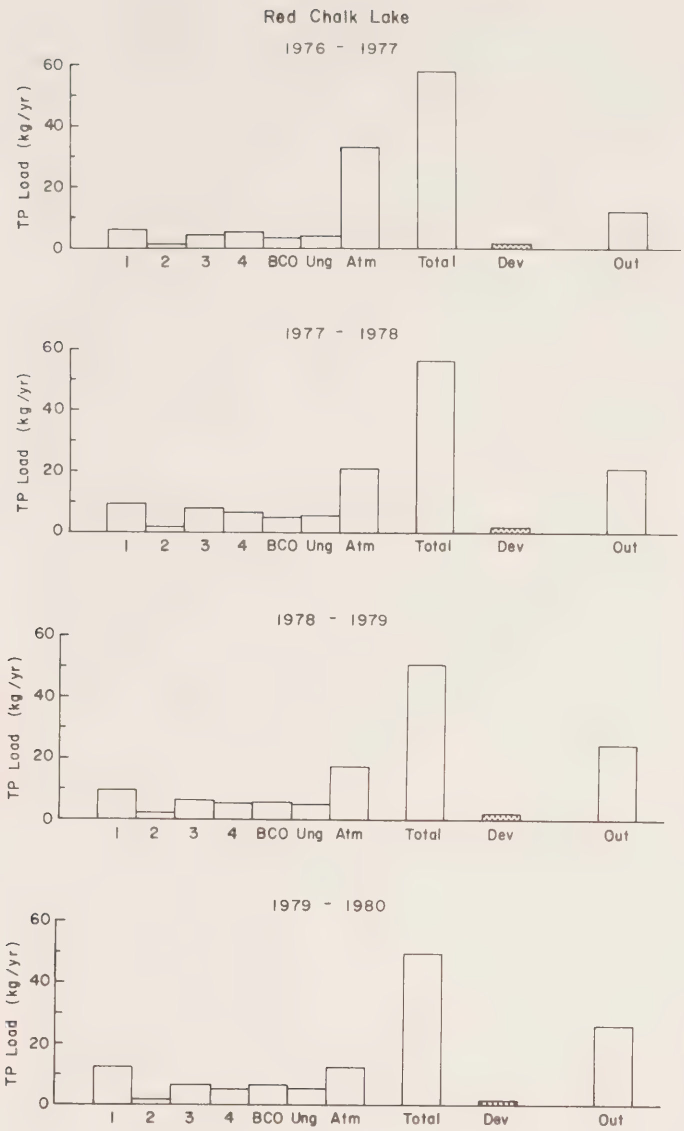


Figure 9 Total phosphorus loading to Red Chalk Lake, in kg/year for four hydrologic years (1976-1980). Inputs are measured streamflow (stream identification number), ungauged area (Ung), atmospheric loading to the lake surface (Atm) and calculated supply from waste disposal systems (Dev). The sum of the natural inputs (Total) and loss by outflow (Out) are also given.

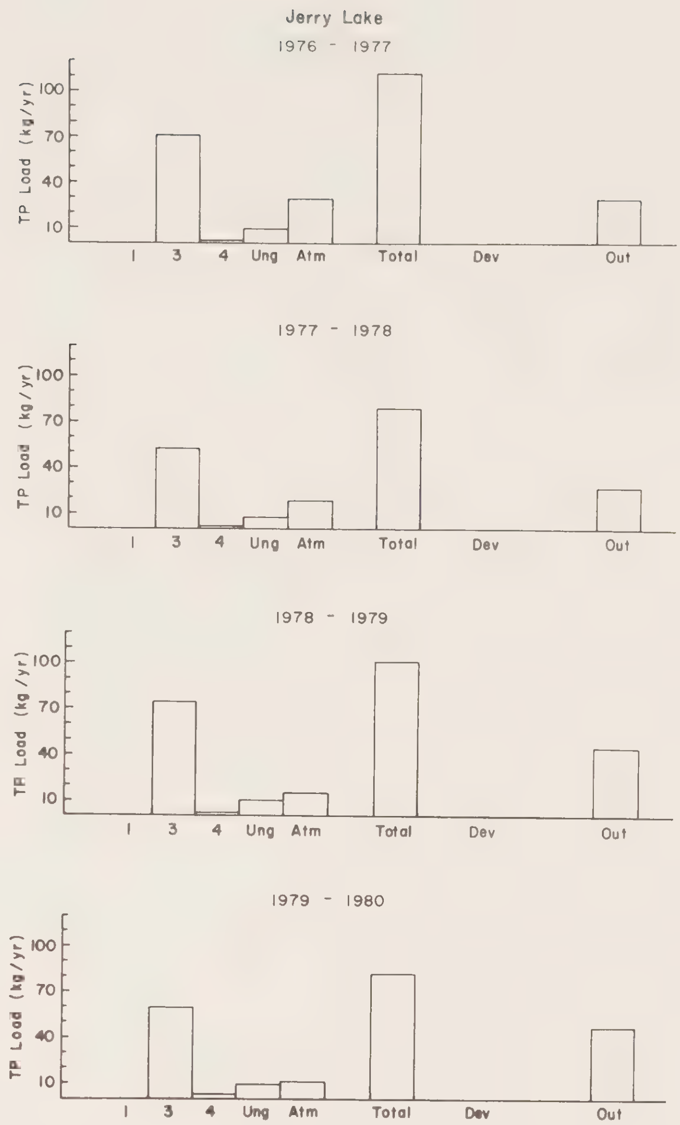


Figure 10 Total phosphorus loading to Jerry Lake, in kg/year for four hydrologic years (1976-1980). Inputs are measured streamflow (stream identification number), ungauged area (Ung), atmospheric loading to the lake surface (Atm) and calculated supply from waste disposal systems (Dev). The sum of the natural inputs (Total) and loss by outflow (Out) are also given.

v) COMPARISON OF THE TOTAL PHOSPHORUS MASS BALANCES

Based on consideration of the measured TP mass balances of the "A" lakes, several generalizations can be made.

The inflowing streams to the study lakes were of highly variable importance, providing as little as 0.4% of the natural input of TP in the case of Jerry Lake inflow 1 or 1% in the case of Harp Lake inflow 3A to as much as 69% (Jerry Lake inflow 3) and 44% (Chub Lake inflow 2).

The input of TP by atmospheric deposition was an extremely important flux; it was the single most important source for four of the six lakes in all years, and the most important in a fifth lake in one of the four years. Atmospheric contributions are most significant for lakes with small terrestrial watersheds.

Contributions of TP from the ungauged watershed areas were not very great, averaging 15% of the total natural input for the six "A" lakes. Therefore, the effect of uncertainty in the measurement of the TP input from these ungauged areas on the TP mass balances is not large.

Loss of TP from the lakes by outflow was always less important than loss by sedimentation, even if the potential anthropogenic inputs were not considered.

Year-to-year variability in individual components of the mass balances was great. The input of TP to each lake by atmospheric deposition varied by almost a factor of three over the four-year study period. Inputs from specific inflows varied by up to a factor of two, as did loss by outflow. The annual inputs of TP to the study lakes from

Table 43. Annual natural total phosphorus load (L_p) to each of the "A" lakes, 4-year average annual load, and coefficient of variation (c.v.) of annual loads for each lake. Potential inputs from lakeshore development are not included.

Lake	L_p (mg m^{-2} yr^{-1})					c.v.
	1976-77	1977-78	1978-79	1979-80	1976-80	(%)
Blue Chalk	68.4	50.2	42.3	36.4	49.3	28
Chub	128.0	112.7	100.9	120.5	115.5	10
Dickie	116.3	93.1	81.2	93.9	96.1	15
Harp	121.1	92.4	101.6	88.8	101.0	14
Jerry	224.2	158.7	204.2	163.1	187.5	17
Red Chalk	101.6	98.8	89.3	87.2	94.2	7
					mean	15

natural sources on an areal basis are summarized in Table 43. The mean annual loads ranged from 49 mg TP m^{-2} yr^{-1} (Blue Chalk Lake) to 188 mg TP m^{-2} yr^{-1} (Jerry Lake). Variability (expressed as the coefficient of variation) in the TP loads ranged from 7% (Red Chalk Lake) to 28% (Blue Chalk Lake), with a mean of 15%. Thus, the 95% confidence interval for the load was equal to the mean load ± 24% (range ± 11% to ± 45%). Results of one-way ANOVA (using z-scored data) indicated that there were highly significant differences in load between years ($p < 0.00002$). The measurement of TP mass balances for several years should be considered essential for development and validation of trophic status models.

In order to establish the importance of the hydrologic balance to the TP loads, the TP loads (Lp) were divided by the areal water load (q_s).

The resulting parameter (L_p/q_s) with units of mg m^{-3} is equivalent to the volume-weighted mean TP concentration of all the measured natural inputs (Table 44). Statistical analysis (one-way ANOVA) indicated that there were still significant differences in L_P/q_s ($p < 0.01$) between years. However, the atmospheric deposition of TP (Section III B-4) was much greater in the first 12-month period (1976-77) than in the subsequent three annual periods. When the values of L_p/q_s for the last three years were compared, there were no significant differences between years ($p > 0.05$). In summary, the volume-weighted mean concentration of TP in the inputs varied only in the one year that atmospheric deposition of TP was very high.

Table 44. Annual total phosphorus load divided by areal water load (L_p/q_s) for each of the "A" lakes. Mean and coefficient of variation are reported. Potential TP inputs from lakeshore development are not included in the calculation.

Lake	L_p/q_s (mg m^{-3})					c.v.
	1976-77	1977-78	1978-79	1979-80	1976-80	(%)
Blue Chalk	52.6	34.9	21.7	17.4	31.7	16
Chub	47.8	26.7	19.8	23.7	29.5	13
Dickie	57.0	39.5	28.1	30.1	38.7	13
Harp	45.7	24.4	19.1	17.5	26.7	13
Jerry	31.3	20.7	18.5	18.3	22.2	6
Red Chalk	23.2	19.1	12.6	12.5	16.8	5
					mean	11

Table 45. Total input of TP to the "A" lakes from natural sources (J_{NAT}), potential input from anthropogenic sources (J_A), ratio of J_A to total potential input calculated from measured mass balances, and original estimate (E) of this ratio from Table 1.

Lake	J_{NAT} (kg yr^{-1})	J_A (kg yr^{-1})	$J_A/(J_A + J_{NAT})$	E
Blue Chalk	24.4	5.7	0.19	0.27
Chub	37.2	2.7	0.07	0.26
Dickie	89.6	77.7	0.46	0.54
Harp	67.6	81.5	0.55	0.61
Jerry	94.0	0.0	0.0	0.0
Red Chalk	53.6	0.6	0.011	0.04

The potential contribution of the anthropogenic sources to the TP mass balances are outlined in Table 45. The results are, except in the case of Chub Lake, very similar to the original estimates (Table 1) made at the start of the study on the basis of existing models. Shoreline development may have a major effect on the trophic status of Harp and Dickie Lakes, an intermediate effect on Blue Chalk Lake, a minor effect on Chub Lake and no effect on Red Chalk or Jerry Lake.

10. THE EFFECTS OF SHORELINE DEVELOPMENT ON THE PHOSPHORUS CONCENTRATIONS OF THE STUDY LAKES

As outlined in Section III.A, the purpose of this section is to evaluate the effects of shoreline development on the trophic status of the "A" lakes. Since the trophic status of a lake is governed by its total phosphorus concentration, this evaluation is based on the contribution of shoreline development to the TP concentrations of the lakes.

As shown in Figure 1, the TP concentration of a lake is a function of the lake's hydrologic budget, TP mass balance and morphometry. Over the past 10-15 years, numerous models (e.g. Vollenweider 1969, Dillon and Rigler 1974, Vollenweider 1975, Reckhow 1977, Walker 1977, and others) have been employed to interrelate these parameters, usually with the explicit purpose of predicting the effects of changes in the input of TP to the lake on its trophic status. In the case of the "A" lakes, the entire input of TP is known for the two undeveloped lakes (Jerry and Red Chalk), but is not known for those with shoreline development. If the concentration of TP in the study lakes is at steady-state (i.e. is not changing – a condition tested in a later section), then a mass balance model can be validated or calibrated using the undeveloped lakes. The validated model may then be applied to the developed lakes to derive a measure of the entire TP input. The difference between the TP input derived in this manner and that part of it that was measured (natural TP input) is attributable to the anthropogenic input of TP, which in turn may be related to the extent of shoreline development.

i) TP CONCENTRATION MODELS

The earliest mass balance models that were used to make predictions about the chemical content of lakes were derived by Vollenweider (1969) and others (reviewed in

Dillon 1974). These models treated lakes as completely mixed reactors, in which chemical inputs were mixed instantaneously throughout the entire water column. Substances were assumed to be supplied at a constant rate from atmospheric deposition, runoff and point sources, and lost at a constant rate via the lake outlet and by sedimentation (Figure 11a). In the simplest case, loss by sedimentation did not occur (i.e. the substance was conservative as outlined in Section III.B.3). However, as discussed in Section III.B.8 and III.B.9, sedimentation from the water column is the major mechanism for loss of TP in the study lakes and must be considered in model formulation.

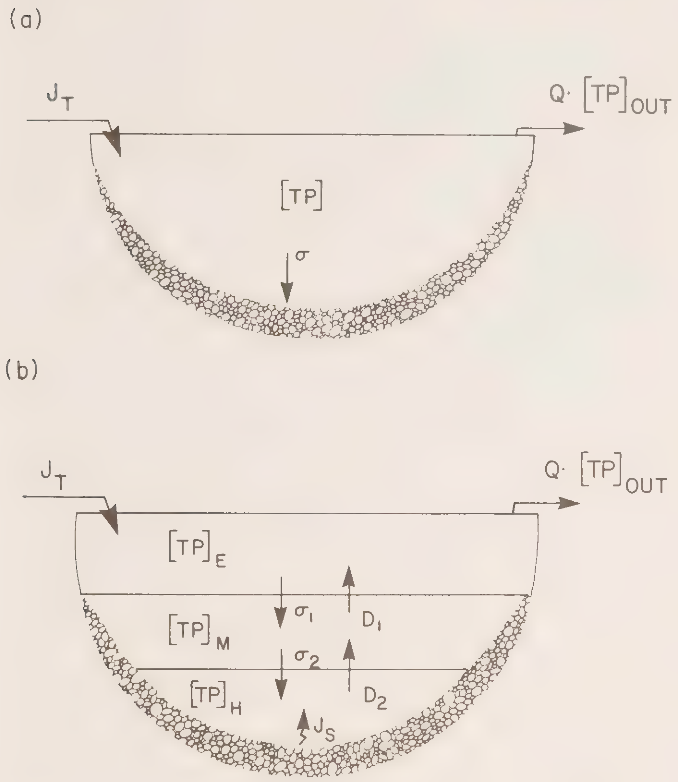


Figure 11 Schematic of TP flux in a lake. In the unstratified model (a), TP is lost by outlow ($Q[TP]_{OUT}$) and by sedimentation (σ). In the "three-box" model (b), TP is transported between layers by diffusion (D_1, D_2) and sedimentation (σ_1, σ_2). Release of TP from the sediments (J_s) may contribute TP to the hypolimnion.

Therefore in the case of TP, this simple model can be expressed in mathematical format as:

$$\frac{d[TP]}{dt} = \frac{J_T}{V} - \frac{Q[TP]_{OUT}}{V} - \sigma\,[TP] \qquad (14)$$

where: $[TP]$ = concentration of TP in the lake at time t
J_T = total input of TP to the lake from external sources per unit time
Q = volume of water flowing out of the lake per unit time
V = lake volume
$[TP]_{OUT}$ = concentration of TP in the lake's outflow
σ = sedimentation rate constant ($time^{-1}$)

If $[TP]_{OUT} = [TP]$, then the concentration of TP at any time, t, is given by:

$$[TP] = J_T\,(1 - e^{-\phi t}) / V\phi + [TP]_o\, e^{-\phi t} \qquad (15)$$

where: $[TP]_o$ = the concentration of TP at t = o
ϕ = the sum of the loss rate constants $= Q/V + \sigma$

At steady-state (i.e. as $t \rightarrow \infty$):

$$[TP]^1 = J_T/V\phi \qquad (16)$$

where: $[TP]^1$ = concentration of TP in the lake at time $t \rightarrow \infty$

Alternatively, loss by sedimentation can be expressed in terms of a sedimentation velocity, v, describing the downward flux of TP:

$$\frac{d[TP]}{dt} = \frac{J_T}{V} - \frac{Q[TP]_{OUT}}{W} - \frac{v.A_s\,[TP]}{} \qquad (17)$$

where: A_s = the area of the plane through which sedimentation occurs
$A_s = A_o$, the lake area, in the simplest case

Thus, $[TP]^1 = J_T / (Q + v.A_o)$ (18)

A third approach to describe the sedimentation of TP is use of the retention coefficient (Dillon and Kirchner 1975b). It can be shown that at steady-state:

$$[TP]^1 = J_T\,(1 - R_p) / Q \qquad (19)$$
$$\text{or } [TP]^1 = L\,(1 - R_p) / q_s \qquad (20)$$

where: R_p = the fraction of the total input of TP not lost via the outflow, and

q_s = areal water load = Q/A_o
L = areal loading rate of TP = J_T/A_o

It can also be shown that equations 17 and 19 are related in the following manner:

$$R_p = v / (v + Q/A_o) \qquad (21)$$
$$\text{or } R_p = v / (v + q_s) \qquad (22)$$

Empirical calibration of equation 22 with data collected for oligotrophic, Precambrian lakes (Dillon and Kirchner 1975) indicated that the settling velocity, v, averaged 12.4 m yr^{-1} (corrected from 13.2 m yr^{-1} in the original paper). Therefore, the simplest version of the TP mass balance model can be written as:

$$[TP]^1 = J_T / (Q + 12.4A_o) \qquad (23)$$
$$\text{or } [TP]^1 = L / (q + 12.4) \qquad (24)$$

In this simple model formulation, the assumption is made that the concentration of TP in the lake is at steady-state. This assumption was tested for the study lakes for the 1976-80 period using TP data (Table 46) collected during five time periods (spring overturn, summer stratified (whole lake) period, fall overturn, ice-free period and annual). For each time period, results of two-way ANOVA showed that there were no significant differences ($p < 0.05$) in TP concentration between years. Therefore, the steady-state version of the model is applicable and the four years can be considered as replicate measurements. This result can be explained on the basis of the response time of the lakes to a change in the input of TP. Since the water replenishment times of the lakes are greater than one year (Table 14), changes in input are not immediately manifested. Measured TP concentrations in the study lakes at spring overturn in the subsequent three years (1980, 1981, 1982) were similar to those measured in the 1976-1980 period (Table 47), confirming that the assumption that the TP concentrations were in steady-state is valid.

Table 46. Total phosphorus concentration (mg m^{-3}) during spring overturn (subscript SO), summer stratification (subscript SS), fall overturn (subscript FO), ice-free (subscript IF), and annual (subscript AN) periods for "A" lakes for each of 4 12-month periods and for 1976-80. Concentrations in the stratified, ice-free and annual periods are whole-lake, volume-weighted averages.

Lake/year	$[TP]_{SO}$	$[TP]_{SS}$	$[TP]_{FO}$	$[TP]_{IF}$	$[TP]_{AN}$
Blue Chalk					
1976-77	4.5	8.1	–[1]	7.8	7.5
1977-78	6.5	7.9	6.6	7.7	7.7
1978-79	6.4	7.0	8.1	7.0	6.8
1979-80	6.9	6.5	7.6	6.5	6.3
1976-80	6.1	7.4	7.4	7.3	7.1
Chub					
1976-77	13.3	11.7	11.6	11.9	11.9
1977-78	13.5	12.2	–[1]	12.4	12.4
1978-79	11.7	12.5	15.1	12.5	12.6
1979-80	15.2	11.6	9.4	11.8	11.8
1976-80	13.4	12.0	12.0	12.2	12.2
Dickie					
1976-77	12.8	10.0	13.2	12.7	13.1
1977-78	13.4	14.4	11.5	13.2	11.8
1978-79	12.4	13.1	14.0	13.2	13.4
1979-80	16.9	14.8	10.7	14.6	13.1
1976-80	13.4	13.1	12.3	13.4	12.8
Harp					
1976-77	7.3	8.1	10.5	8.1	8.0
1977-78	6.7	8.4	9.2	8.2	8.1
1978-79	8.6	7.2	8.7	7.5	7.4
1979-80	8.5	7.5	6.8	7.5	7.7
1976-80	7.8	7.8	8.8	7.8	7.8
Jerry					
1976-77	–[1]	10.5	9.9	10.5	9.8
1977-78	7.7	9.9	9.7	9.7	9.5
1978-79	6.9	8.5	8.0	8.3	8.5
1979-80	8.6	8.8	9.5	8.9	8.8
1976-80	7.7	9.4	9.3	9.3	9.1
Red Chalk					
1976-77	5.0	5.3	6.4	5.4	5.7
1977-78	6.8	6.4	5.1	6.4	6.1
1978-79	4.7	5.4	7.4	5.5	5.6
1979-80	5.4	5.1	4.8	5.3	5.4
1976-80	5.5	5.6	5.9	5.6	5.7

[1] no data

Table 47. TP concentration at spring overturn from 1980 to 1982. All results in mg m^{-3}.

Lake	1980	1981	1982
Blue Chalk	6.5	6.3	6.7
Chub	12.6	12.4	10.6
Dickie	11.5	12.0	9.5
Harp	8.3	7.0	7.0
Jerry	8.1	–[1]	–[1]
Red Chalk	4.8	4.6	4.9

[1] no data

ii) IMPROVEMENTS IN TP CONCENTRATION MODELS

Many alterations to the simple mass balance model have been proposed in the past decade. Changes can be classified as being of two general types:

a) those that are based on a more "realistic" concept of a lake. (Figure 11b). For example, some of these consider thermal stratification (resulting in a 2- or 3-layer model for part of the year), TP release from the sediments to the water column, specific fractions of the TP pool (e.g. particulate phosphorus and dissolved phosphorus), and other modifications;

b) those that evaluate model parameters and coefficients by statistical techniques such that the resulting model is empirical or semi-empirical rather than a true mass balance model (eg. Reckhow 1977, Walker 1977).

Some of these changes are considered in this section but the approach taken is to use the simplest model that makes accurate predictions. This approach was followed principally because the simplest models generally require the least measurements. Improvements based on a more "realistic" conceptual model of a lake are discussed first.

In instances where input parameters are expressed on an annual basis, the time period for which the prediction of TP concentration is applied has been considered to be the spring overturn in some cases (Dillon and Rigler 1974a), the annual average in others and the ice-free period average in yet other instances. Although empirical models may be tested or calibrated on any temporal basis, mass balance models which are semi-empirical (i.e. contain some empirical and some non-empirical steps) or non-empirical may best predict annual TP concentrations or at least TP concentrations representing most of the year. Since it is desired to predict trophic status parameters (chlorophyll *a*, phytoplankton biomass, Secchi disc depth) during the ice-free period on the basis of TP concentration, the model was applied to predict mean ice-free values of TP concentration. The difficulty of obtaining good estimates of mean annual TP concentration, because of the problems encountered in sampling in winter, reinforces the decision to use averages for the ice-free period. To investigate further the consequences of this decision, the differences in TP concentration between seasonal time periods in the study lakes were tested (data summarized in Table 46). There were no significant differences (paired t-test, $p > 0.05$) between the mean TP concentrations of the ice-free period and those of the spring overturn, summer stratified, fall overturn or annual periods, nor were there any significant differences between values measured in the spring overturn and any other time period. These results suggest that seasonal patterns in the TP concentrations in the study lakes were not pronounced. However, highly significant correlations between the TP concentrations measured in different time periods were found. The relationships between TP concentrations in the ice-free period and those in four other time periods (spring overturn, summer stratified, fall overturn, annual) are given in Table 48. These relationships allow estimates of TP concentration for these periods to be made from the predictions made by the ice-

free model, or estimates of ice-free concentration to be made from measurements made during other time periods.

Differences in TP concentration in vertical depth zones resulting from thermal stratification were observed in some study lakes (Table 49). The TP concentrations in the hypolimnia of Blue Chalk, Chub and Dickie Lakes (in years when it was thermally stratified) were much greater than concentrations measured in the epilimnia of the same lakes (paired t-test, $p < 0.05$), with differences of 5.9-8.1 mg m^{-3}. The differences in Harp, Jerry and the main basin of Red Chalk Lake were much less, ranging from only 0.6-3.2 mg m^{-3}. Mass balance models incorporating thermal stratification during part of the year have been proposed and utilized (Snodgrass and O'Melia 1975). Snodgrass and Dillon (1983) compared the predictive abilities of one and two-layer models using a data set collected in the Muskoka-Haliburton study area (Dillon 1974). The one-layer model, similar in structure to that proposed in equation 15, provided equally accurate predictions of TP concentrations as did the two-layer model (an epilimnion and a hypolimnion with the thermocline treated as a two-dimensional plane). Therefore, it is concluded that it is unneccessary to modify the mass balance model to account for thermal stratification.

Table 48. Relationships between TP concentration (mg m^{-3}) in the ice-free period and that measured in 4 other time periods. Results for each time period were averaged over the 4 years for each lake.

A. Predicting ice-free period concentration	
ice-free vs. spring overturn	
$[TP]_{IF} = 0.80\,[TP]_{SO} + 2.04$	$r^2 = 0.943$
ice-free vs. fall overturn	
$[TP]_{IF} = 1.04\,[TP]_{FO} - 0.15$	$r^2 = 0.913$
ice-free vs. summer stratified	
$[TP]_{IF} = 1.04\,[TP]_{SS} - 0.33$	$r^2 = 0.998$
ice-free vs. annual	
$[TP]_{IF} = 1.04\,[TP]_{AN} - 0.23$	$r^2 = 0.996$
B. Predicting other concentrations from ice-free concentration	
spring overturn vs. ice-free	
$[TP]_{SO} = 1.18\,[TP]_{IF} - 1.91$	$r^2 = 0.943$
fall overturn vs. ice-free	
$[TP]_{FO} = 0.88\,[TP]_{IF} + 0.91$	$r^2 = 0.913$
summer stratified vs. ice-free	
$[TP]_{SS} = 0.96\,[TP]_{IF} + 0.33$	$r^2 = 0.998$
annual vs. ice-free	
$[TP]_{AN} = 0.96\,[TP]_{IF} + 0.25$	$r^2 = 0.996$
epilimnetic vs. ice-free	
$[TP]_{EP} = 0.95\,[TP]_{IF} - 1.23$	$r^2 = 0.903$
epi- and metalimnetic vs. ice-free	
$[TP]_{EM} = 0.90\,[TP]_{IF} + 0.09$	$r^2 = 0.884$
C. Predictions based on summer stratification concentration	
epilimnetic vs. summer stratification	
$[TP]_{EP} = 0.98\,[TP]_{SS} - 1.44$	$r^2 = 0.881$
epi- and metalimnetic vs. summer stratification	
$[TP]_{EM} = 0.93\,[TP]_{SS} - 0.11$	$r^2 = 0.863$

Table 49. Total phosphorus concentration (mg m^{-3}) in the epilimnion (subscript E), epi- and metalimnion (subscript EM), hypolimnion (subscript H) and in the whole lake during the summer stratification (subscript SS) period for "A" lakes for each of 4 12-month periods and for 1976-80. Concentrations in the epi- and metalimnion and during summer stratification are volume-weighted averages.

Lake/year	$[TP]_E$	$[TP]_{EM}$	$[TP]_H$	$[TP]_{SS}$
Blue Chalk				
1976-77	5.6	7.1	14.1	8.1
1977-78	6.3	6.9	16.4	7.9
1978-79	5.6	6.3	12.6	7.0
1979-80	5.1	5.7	11.7	6.5
1976-80	5.7	6.5	13.7	7.4
Chub				
1976-77	9.1	9.3	16.5	11.7
1977-78	8.9	9.4	19.6	12.2
1978-79	10.6	10.7	16.9	12.5
1979-80	8.7	9.0	16.8	11.6
1976-80	9.3	9.6	17.4	12.0
Dickie				
1976-77	13.5	14.1	–[1]	10.0
1977-78	11.4	13.1	17.8	14.4
1978-79	12.1	12.7	17.9	13.1
1979-80	13.7	14.2	20.0	14.8
1976-80	12.7	13.5	18.6	13.1
Harp				
1976-77	7.3	7.9	8.5	8.1
1977-78	7.5	8.2	8.9	8.4
1978-79	6.2	7.0	7.9	7.2
1979-80	6.4	7.3	8.0	7.5
1976-80	6.9	7.6	8.3	7.8
Jerry				
1976-77	6.5	8.3	13.6	10.5
1977-78	7.5	8.5	12.5	9.9
1978-79	7.3	8.0	9.8	8.5
1979-80	7.8	8.0	10.3	8.8
1976-80	7.3	8.2	9.3	9.4
Red Chalk				
1976-77	4.5	5.6	5.2	5.3
1977-78	5.5	6.6	6.3	6.4
1978-79	5.0	5.7	5.1	5.4
1979-80	4.4	5.1	5.2	5.1
1976-80	4.9	5.7	5.5	5.6

[1] no hypolimnion

The assumption that the TP concentration in the lake outlet is equal to the concentration in the lake is inherent in the simple mass balance model. Based on earlier published studies, Dillon et al. (1978) suggested that the relationship in TP concentration between lake outlet and lake concentrations could be approximated as:

$$[TP]_{OUT} = 0.9\,[TP]_{AN} \qquad (25)$$

where: $[TP]_{AN}$ = mean annual TP concentration in the lake.

The mean annual TP concentrations of the outflows of the study lakes are reported in Table 50. The relationship between these concentrations and the mean ice-free period lake concentrations is:

$$[TP]_{OUT} = 0.956\,TP_{IF}\ (r^2 = 0.80) \qquad (26)$$

If this relationship is considered in the model derivation described in equations 17 and 24, the resulting solution is:

$$[TP]^1 = L\,(1 - R_p) / 0.956\, q_s \tag{27}$$

$$\text{or } [TP]^1 = L / (0.956\, q_s + 12.4) \tag{28}$$

In subsequent calculations, the model defined by equation 28 is termed "LCS-1".

An alternate method of deriving the loss of TP in a lake by sedimentation was outlined in Section III.B.8. Site-specific whole-lake Pb burdens were used to pro-rate measured sediment accumulation rates (Pb-210 methodology), which were combined with the measured TP content of the sediments to estimate the long-term whole-lake TP sedimentation rate (S).

The mass balance model can therefore be re-formulated as:

$$[TP]^1 = (L - S) / 0.956\, q_s \tag{29}$$

This model is subsequently described as "LCS-2".

Other advances in TP model development have been based on statistical techniques. Model coefficients have been empirically derived by fitting measured data to the basic mass balance model. For example, Reckhow (1977) used a global data set to develop the relationship:

$$[TP] = L / (11.6 + 1.2\, q_s) \tag{30}$$

where the 11.6 (equivalent to the settling velocity) and the 1.2 were estimated using statistical optimization techniques. Using the same techniques but an expanded data set, Walker (1977) proposed an alternative model:

$$[TP] = L / q_s\,(1 + 0.824\, \tau_w^{0.454}) \tag{31}$$

Table 50. Mean annual TP concentration (mg m^{-3}) in the outflow of the study lakes.

Lake	1976-77	1977-78	1978-79	1979-80
Blue Chalk	7.33	9.06	9.86	8.15
Chub	11.1	10.7	9.77	10.5
Dickie	12.0	13.0	12.5	12.4
Harp	9.36	7.52	7.88	8.93
Jerry	8.27	8.48	7.63	9.53
Red Chalk	9.42	7.04	5.99	7.08

Table 51. Empirically derived predictive mass balance models. These models were developed using statistical techniques and are formulated on the basis of best fit to global data sets.

Source	Model	Conditions
Reckhow (1977)	$[TP] = L/(11.6 + 1.2\, q_s)$	general
Walker (1977)	$[TP] = L/q_s(1 + 0.824\, \tau_w^{0.454})$	general
Reckhow (1977)	$[TP] = L/((18\bar{z}/(10 + \bar{z}) + 1.05\, q_s \exp(0.012\, q_s))$	oxic lakes, $q_s < 50$ m yr^{-1}
Reckhow (1977)	$[TP] = L/(0.17\bar{z} + 1.13\, q_s)$	anoxic lakes

Significant improvements in predictive capability were achieved by Reckhow (1977), when he divided the lake set into those with completely oxic hypolimnia throughout the period of stratification and those with partially or totally anoxic hypolimnia. Only those lakes with q_s <50m yr^{-1} were considered in the oxic subset in his analysis.

These results (summarized in Table 51) indicate that there are inherent differences in the way in which lakes with anoxic as opposed to oxic hypolimnia process TP. This finding is consistent with the observation that the study lakes with anoxic hypolimnia had greater differences between hypolimnetic and epilimnetic TP concentration than did those that had oxic hypolimnia. These individual models were also tested using the data collected for the study lakes.

iii) MODEL CALIBRATION – JERRY LAKE

The previous sections outlined the rationale for use of the TP mass balance lake models in a form based on the following assumptions:

a) the TP concentration of the lake is in steady-state,
b) the model predicts the mean TP concentration in the ice-free period, but it can be related to the TP concentration in other periods,
c) the input data to the model are averages of four years (1976-1980) of data,
d) the TP inputs to the lake are mixed throughout the water column at all times; i.e. the one-layer version of the model (no thermal stratification) is suitable,
e) the concentration of TP in the lake outlet is related to the concentration of TP in the lake by the relationship shown in equation 26.

There was no potential contribution of TP from shoreline development to Jerry Lake (Table A18), and the potential input to Red Chalk Lake (Table A17) was close enough to zero 50 as to be insignificant (potential anthropogenic input = 1.1% total natural input). Therefore, both lakes can be used to validate the TP mass balance lake model.

Except for the sedimentation rate (or retention coefficient), the information needed to predict the TP concentration of Jerry Lake using the model designated as LCS-1 is provided in Tables 14 and A18. Since J_A = O and since the TP concentration is in steady-state, the measured retention represents the actual loss of TP to the lake's sediments. However, this value of retention cannot be used for prediction purposes since it is not measured independently of the TP budget.

The retention coefficients predicted on the basis of the model outlined in equation 22 (with settling velocity v = 12.4 m yr^{-1}) are compared to measured retention coefficients in Table 52. As expected, the long-term (1976-1980) average measured and predicted retention coefficients are in much better agreement than the individual annual values. This occurs because the lake response times are >1 year and changes in annual input of TP are

Table 52. Measured (by mass balances) and predicted TP retention (R_p) in Jerry Lake. Predicted retention calculated using settling velocity of 12.4 m yr^{-1}.

	R_p				
	1976-77	1977-78	1978-79	1979-80	1976-80
Measured	0.734	0.652	0.551	0.420	0.598
Predicted	0.634	0.618	0.529	0.582	0.588

therefore not necessarily reflected in corresponding changes in output (and therefore retention) in the same year. If the response time of the lake was rapid (<1 year), then better agreement on an annual basis would be expected. Conversely, lakes with water replenishment rates of decades would require even longer-term data than those collected in this study.

Since the prediction of TP concentration using the mass balance model (equation 27) is directly proportional to the term ($1-R_p$), comparison of predicted and measured retention is, in effect, a test of the predictive ability of the mass balance model. Nevertheless, the TP mass balance model can be validated in a more conventional manner by comparing predicted vs observed concentrations of TP. The calculations of the predicted TP concentration are shown in Table 53 using a value of retention predicted on the basis of a settling velocity of 12.4 m yr^{-1}. The predicted steady-state TP concentration (9.3 mg m^{-3}) is identical to the average concentration measured in the ice-free period (Table 46). Therefore, the observed TP concentration can be adequately explained by the measured TP input and that the retention model is satisfactory.

Table 53. Prediction of TP concentration in Jerry Lake using LCS-1 model (equations 27 and 32)

Parameter	Source	4-year average
L	Table 61	187.5 mg m^{-2} yr^{-1}
q_s	Table 20	8.69 m yr^{-1}
R_p	Table 70	0.588
[TP]	$\frac{L\,(1-R_p)}{0.956\,q_s}$	9.3 mg m^{-3}
$[TP]_{MEAS}$	Table 64	9.3 mg m^{-3}

An alternative calculation (LCS-2) can be made using the estimated long-term accumulation of TP in the sediments of Jerry Lake (99 mg TP m^{-2} yr^{-1} calculated from the Pb-210 method (Table 40)) to calculate the retention coefficient. This method yields a prediction of [TP] = 10.6 mg m^{-3}, somewhat higher than the measured [TP] of 9.3 mg m^{-3}.

iv) MODEL CALIBRATION – RED CHALK LAKE

As previously discussed, Red Chalk Lake data can be used as an additional validation of the TP mass balance model because the potential anthropogenic TP input is insignificant. If the measured and predicted (using settling velocity v = 12.4 m yr^{-1}) TP retention coefficients are compared (Table 54) based on 1976-1980 data, it is apparent that the predicted retention is greater than the measured. There is substantial variability between individual years, with the measured retention exceeding the predicted in one of the four years.

Calculations based on the mass balance model (Table 55) yield a predicted TP concentration of 5.4 mg m^{-3}, which is within experimental error of the measured (Table 46) value[1] of 5.6 mg m^{-3}. Despite this good agreement, it was believed that the small eastern basin of Red Chalk Lake may act as a disproportionately important sink for TP since most of the TP input passes into this basin before the main basin. This hypothesis is supported by the observation (Table 56) that the TP concentration in the east basin was substantially greater than that in the main basin during the ice-free period (7.8 mg m^{-3} vs 5.6 mg m^{-3}). Further comparisons of the TP concentrations in the east basin and main basin at spring overturn (7.4 mg m^{-3} vs 5.5 mg m^{-3} respectively), fall overturn (6.3 mg m^{-3} vs 5.9 mg m^{-3} respectively) and at other time periods demonstrated that the TP concentration of the east basin was always greater. However, by far the greatest differential in TP concentrations between the two basins was in the hypolimnetic TP values (18.0 mg m^{-3} vs 5.5 mg m^{-3} respectively).

Table 54. Measured (by mass balances) and predicted TP retention in Chalk Lake. Predicted retention calculated using settling velocity of 12.4 m yr^{-1}.

	R_p				
	1976-77	1977-78	1978-79	1979-80	1976-80
Measured	0.789	0.630	0.516	0.470	0.608
Predicted	0.739	0.706	0.636	0.640	0.677

Table 55. Prediction of TP concentration in Red Chalk Lake using LCS-1 model (equations 27 and 32)

Parameter	Source	4-year average
L	Table 61	94.2 mg m^{-2} yr^{-1}
q_s	Table 20	5.91 m yr^{-1}
R_p	Table 72	0.677
[TP]	$\frac{L\,(1-R_p)}{0.956\,q_s}$	5.4 mg m^{-3}
$[TP]_{MEAS}$	Table 64	5.6 mg m^{-3} (5.8)[1]

[1] see footnote on page

In order to clarify the causes of these differences, hydrologic and TP mass balances were constructed separately for each basin. The transfer of water from the east basin to the main basin was calculated as the residual term in the water balance (all other terms measured) for Red Chalk East (Table 57) since no direct measurements were available. This estimate was corrected to account for the error observed in the Red Chalk (whole lake) water balance (Table A6) by apportioning the error term on the basis of the ratio of the watershed area of Red Chalk East to Red Chalk (whole basin). The areal water load (q_s) was calculated for Red Chalk East as the estimated outlet Q/lake area of Red Chalk East. Similarly, the areal water load of the main basin was calculated as outlet Q/area main basin (Table 58). A measure of the transport of TP from the east basin to the main basin was derived using the estimated outflow from the east basin and the TP concentration in the basin. Using this information and

[1] The mean TP concentration of 5.6 mg m^{-3} was calculated from measurements taken at the main basin of Red Chalk Lake. Calculating the mean using measurements taken at both the main and east basins (weighted to account for differences in volume of water between the basins), a value of 5.8 mg m^{-3} is obtained.

Table 56. Comparison of total phosphorus concentrations (mg m^{-3}) in the main basin and east basin of Red Chalk Lake. Four annual averages and the 4-year average are reported for different seasons and different depth zones (defined in Tables 46 and 49).

	$[TP]_{SO}$	$[TP]_E$	$[TP]_{EM}$	$[TP]_H$	$[TP]_{SS}$	$[TP]_{IF}$	$[TP]_{FO}$	$[TP]_{AN}$
Main basin								
1976-77	5.0	4.5	5.6	5.2	5.3	5.4	6.4	5.7
1977-78	6.8	5.5	6.6	6.3	6.4	6.4	5.1	6.1
1978-79	4.7	5.0	5.7	5.1	5.4	5.5	7.4	5.6
1979-80	5.4	4.4	5.1	5.2	5.1	5.3	4.8	5.4
1976-80	5.5	4.9	5.7	5.5	5.6	5.6	5.9	5.7
East basin								
1976-77	–	–	–	–	–	–	–	–
1977-78	7.5	5.5	7.4	20.5	8.7	8.6	7.4	8.5
1978-79	6.7	5.0	6.1	16.2	7.5	7.4	5.5	7.2
1979-80	8.1	4.7	5.8	17.1	7.4	7.3	6.0	7.1
1976-80	7.4	5.1	6.5	18.0	7.9	7.8	6.3	7.6

other data from Table A17, the TP mass balances for the east basin and main basin of Red Chalk Lake were constructed (Tables 59 and 60 respectively).

The predicted concentration of TP in each of the basins is shown in Table 61. In the main basin, the predicted and observed concentrations (ice-free period) were almost identical (5.5 mg m^{-3} vs 5.6 mg m^{-3} respectively), whereas in the east basin the predicted concentration was substantially lower than the observed (6.4 mg m^{-3} vs 7.8 mg m^{-3} respectively). The TP mass balance model therefore provided accurate predictions of TP concentration in Jerry and Red Chalk (Main basin) Lakes but not in Red Chalk East. The difference between these lakes was that the hypolimnia of Jerry and Red Chalk Main were oxic throughout the period of summer stratification while the hypolimnion of Red Chalk East was not (Figure 12). A manifestation of this was that the TP concentration in the hypolimnion of the east basin of Red Chalk Lake was much greater than that in the epilimnion (Table 56) whereas this was not the case in the main basin of Red Chalk Lake or in Jerry Lake (Table 49).

Table 57. Water balance for Red Chalk Lake East Basin. All results in 10^6 m^3 yr^{-1} with the exception of q_s (m yr^{-1}) and R_p (calculated using equation 22).

Source	1976-77	1977-78	1978-79	1979-80	mean
INPUTS					
Blue Chalk Outflow	0.644	0.713	0.962	1.030	0.837
Inflow 1	0.648	0.695	0.872	0.885	0.775
Inflow 2	0.094	0.120	0.175	0.149	0.135
Precipitation	0.107	0.106	0.176	0.153	0.136
Ungauged	0.108	0.140	0.170	0.160	0.145
Total	1.601	1.774	2.355	2.377	2.027
OUTPUTS					
Evaporation	0.093	0.075	0.089	0.085	0.086
Storage	0.000	+0.015	+0.004	–0.003	+0.004
Outflow	1.508	1.684	2.262	2.295	1.937
Corrected outflow[1]	1.651	1.814	2.491	2.527	2.121
q_s	12.7	14.0	19.2	19.4	16.3
R_p	0.494	0.470	0.392	0.390	0.432

[1] error in the whole-lake water balance divided between the East and Main Basins on the basis of relative watershed and basin areas.

Table 58. Water balance for Red Chalk Lake Main Basin. Outflow is expressed in 10^6 m^3 yr^{-1}; q_s in m yr^{-1}. R_p is calculated from equation 22.

Source	1976-77	1977-78	1978-79	1979-80	mean
Outflow	2.490	2.940	4.040	3.970	3.360
q_s	5.67	6.70	9.20	9.04	7.65
R_p	0.686	0.649	0.574	0.578	0.622

Table 59. Total phosphorus mass balance for Red Chalk Lake East Basin. All results in kg yr^{-1}, except load (mg m^{-2} yr^{-1}).

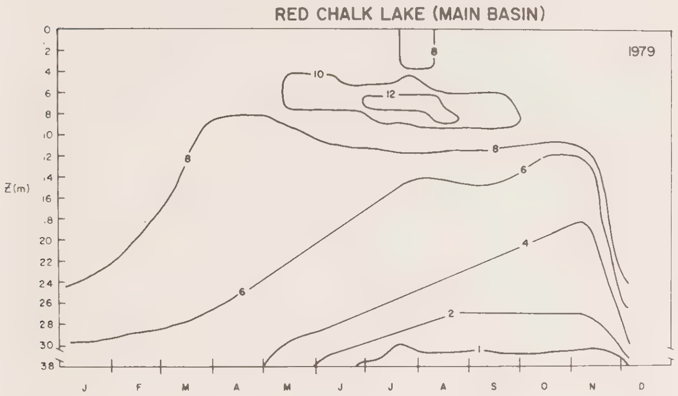
Source	1976-77	1977-78	1978-79	1979-80	mean
Blue Chalk Outflow	3.49	4.86	5.57	6.51	5.11
Inflow 1	6.23	9.32	9.58	12.50	9.41
Inflow 2	1.31	1.69	2.01	1.67	1.67
Precipitation	7.59	4.73	3.93	2.83	4.77
Ungauged	1.46	2.02	1.83	2.01	1.83
Total	20.08	22.62	22.92	25.52	22.79
Load	154.5	174.0	176.3	196.3	175.3

Table 60. Total phosphorus mass balance for Red Chalk Lake Main Basin. All results in kg yr^{-1}, except load (mg m^{-2} yr^{-1}).

Source	1976-77	1977-78	1978-79	1979-80	mean
Red Chalk East	12.88	15.60	18.43	18.45	16.34
Inflow 3	4.31	7.83	6.44	6.20	6.20
Inflow 4	5.32	6.41	5.19	4.93	5.46
Precipitation	25.61	15.97	13.27	9.57	16.11
Ungauged	2.42	3.33	3.01	3.31	3.02
Total	50.54	49.14	46.34	42.46	47.12
Load	115.1	111.9	105.6	96.7	107.3

As mentioned previously in this section, Reckhow (1977) proposed that different mass balance models were required for oxic and anoxic lakes. In Table 62, predictions of TP concentration made on the basis of Reckhow's oxic and anoxic models, the Reckhow general model, the Walker (1977) general model and the LCS-1 and LCS-2 models (equations 28 and 29) derived in this study are compared to the measured TP concentrations in Jerry

Lake and the two basins (plus the whole lake) of Red Chalk Lake. For the oxic lakes, the LCS-1 model makes the best predictions, although the Reckhow oxic model is almost as good. The Reckhow general model, which makes no distinction between oxic and anoxic lakes, underestimated the TP concentrations, whereas the Walker model overestimated them. The LCS-2 model made poorer predictions than LCS-1, and was excluded from subsequent calculations. However, none of the models provided an accurate prediction of the TP concentration of the anoxic east basin of Red Chalk Lake.



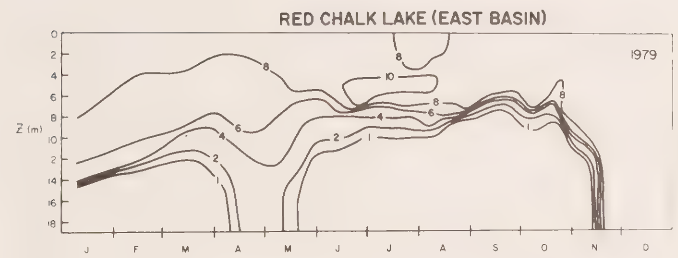


Figure 12 Dissolved oxygen (mg L^{-1}) isopleth maps for Red Chalk Main and Red Chalk East Basin for 1979.

On the basis of these findings, it was decided that an alternate formulation of the mass balance model for anoxic lakes was required. Increased TP concentration in the hypolimnion of the east basin indicates either release of TP from the sediments or less efficient removal from the water column. These two possibilities are indistinguishable, and can be reported mathematically by a decrease in the settling velocity. Since the anthropogenic input of TP from lakeshore development can make no contribution to the TP mass balance or the TP concentration of the east basin of Red Chalk Lake, the LCS-1 model can simply be calibrated by optimizing the settling velocity (v) such that the predicted and observed TP concentrations agree. The result of this procedure is that for anoxic lakes the proposed settling velocity is 7.2 m yr^{-1}, as opposed to 12.4 m yr^{-1} for oxic lakes. The LCS model therefore has the form:

$$[TP]^1 = L(1 - R_p) / 0.956\, q_s \qquad (27)$$

where:

$$R_p = 12.4 / (12.4 + q_s) \text{ for oxic lakes} \qquad (32)$$

$$R_p = 7.2 / (7.2 + q_s) \text{ for anoxic lakes} \qquad (33)$$

Table 61. Prediction of TP concentration (mg m^{-3}) in each basin of Red Chalk Lake.

	Main basin		East basin	
Parameter	Source	4-year average	Source	4-year average
L (mg m^{-2} yr^{-1})	Table 78	107.3	Table 77	175.3
q (m yr^{-1})	Table 76	7.65	Table 75	16.3
R_p	Table 76	0.622	Table 75	0.432
[TP] (mg m^{-3})	$\frac{L(1-R_p)}{0.956q_s}$	5.5(4)	$\frac{L(1-R_p)}{0.956q_s}$	6.4
$[TP]_{MEAS}$ (mg m^{-3})	Table 74	5.6	Table 74	7.8

Table 62. Comparison of predicted and measured TP concentrations (mg m^{-3}) for Jerry and Red Chalk Lakes. Predictions are based on 4 models reported in the literature and the LCS models resulting from this study.

		Red Chalk		
	Jerry	Whole	Main	East
PREDICTED				
Reckhow – general	8.5	5.0	5.2	5.6
Reckhow – oxic	9.3	5.5	5.3	–
Reckhow – anoxic	–	10.4	–	9.0
Walker – general	11.0	7.2	6.5	7.1
LCS – 1	9.3	5.4	5.5(4)	6.4
LCS – 2	10.6	1.6	2.3	6.2
MEASURED	9.3	5.6[1]	5.6	7.8

[1] measured in Main basin.

v) EFFECTS OF LAKESHORE DEVELOPMENT

The potential contribution of anthropogenic TP from lakeshore development to the TP mass balances and TP concentrations of Chub, Blue Chalk, Dickie and Harp Lake is significant (Table 45). In order to estimate the proportion of this potential TP input that actually entered the lake, the model developed and calibrated in the previous section was employed.

The potential contribution of anthropogenic TP to the TP supply of Chub and Blue Chalk Lake was originally estimated to be 26% and 27% of the total input respectively (Table 1), based on estimated natural inputs and number of dwelling units. However, measurements made as part of this study (Tables 32, A15, A17) indicated that the revised potential contribution of anthropogenic TP was only 7% and 19% of the total for Chub and Blue Chalk Lakes respectively (Table 45). The effect of this input on Chub Lake could only be small.

Both Chub and Blue Chalk Lake had anoxic hypolimnetic waters during late summer and in the fall prior to destratification (Reid et al. 1983a). Therefore the anoxic models (equations 27 and 33) are suitable for these lakes. The TP concentration predicted on the basis of the measured natural TP inputs was less than the measured concentration (Table 63) in both lakes. The difference in Chub Lake was 1.8 mg m^{-3} whereas it was 1.5 mg m^{-3} in Blue Chalk Lake. Although these differences are not great, they are both in excess of the estimated analytical errors

in the measurements (0.5 mg m^{-3}) and much greater than the differences between predicted and observed TP concentrations in the case of Red Chalk Lake (Main basin)[1] and Jerry Lake (predicted TP concentration agreed with observed to within 0.1 mg m^{-3} in both lakes). If the entire potential contribution of anthropogenic TP from lakeshore development (L_A in Table 63) is included in the TP mass balance, then the predicted and observed TP concentrations in the lakes are in much better agreement, differing by 1.0 mg m^{-3} in Chub Lake and 0.1 mg m^{-3} in Blue Chalk Lake. In each case, the measured TP concentration is still greater than the predicted concentration, although the difference for Blue Chalk Lake is insignificant. The results indicate that shoreline development is contributing to the TP concentration in these lakes. In fact, the results suggested that additional inputs of TP to Chub Lake occurred. Although measurement or model error could explain this underestimate, it is also possible that the data on human use (capita years yr^{-1}) of the lake are in error. Survey information (Table 31) suggested human use of Chub Lake to be 4.01 capita years yr^{-1} in 1976. Estimating human use by the "accessibility equation" (Downing 1983), the "cottager equation" (Downing 1983) and the "average equation" (Downing 1983, using Blue Chalk Lake as a benchmark lake) gives values of 8.4 cap yrs yr^{-1}, 6.6 cap yrs yr^{-1} and 6.4 cap yrs yr^{-1} respectively. Estimates of human use for Chub Lake in 1978 obtained by employing the latter three methods (9.2, 7.3 and 7.1 cap yrs yr^{-1}) were also considerably higher than the estimate obtained from survey information (4.4 cap yrs yr^{-1}). If the average value of human use calculated from the three equations outlined in Downing (1983) is used, a potential anthropogenic TP input of 15.1 mg m^{-2} yr^{-1} is obtained. This, in turn, results in a predicted TP concentration of 11.8 mg m^{-3}, in close agreement with the observed value of 12.2 mg m^{-3}.

Original estimates (Table 1) indicated that the potential contributions of anthropogenic TP from shoreline development to the TP mass balances of Harp Lake and Dickie Lake were 61% and 54% respectively. Although these figures were reduced (to 55% and 46% respectively; Table 45) when measured data on natural inputs and human use were collected, the potential contribution of the anthropogenic TP remained high.

The hypolimnion of Harp Lake remained oxygenated at all times allowing use of the oxic model (equations 27 and 32). However, Dickie Lake did not have a hypolimnion in all years of study (Reid et al. 1983a), undoubtedly because of its small maximum depth (12m) and relatively large surface area (≃0.93 km^2). Dickie Lake was only weakly stratified in all years, with two discernible layers in the summer of 1976 and three in the subsequent three summers. The oxygen content of the bottom waters was low (<1 mg L^{-1}) in each of the years, but the bottom waters remained oxic (Figure 13). However, in two of the three subsequent summers (1981, 1982), the bottom waters were anoxic. The calculations were therefore carried out for Dickie Lake using both the anoxic and oxic models, although the information gathered indicates that the oxic model was more appropriate.

The TP concentration predicted for Harp Lake using the measured natural load (101 mg TP m^{-2} yr^{-1}) was less than the observed concentration (6.4 vs 7.8 mg m^{-3}; Table 64). However, if all of the potential anthropogenic TP input from shoreline development reached the lake, the expected concentration (12.8 mg m^{-3}) exceeded the measured concentration by a considerable margin. The mass balance model can be used to estimate the anthropogenic TP input by equating the observed TP concentration with the model formulation. If this is done, the total load must have been 123.8 mg TP m^{-2} yr^{-1}. Therefore, the actual input from shoreline development (22.8 mg m^{-2} yr^{-1}) represents 19% of the potential contribution.

The predicted TP concentration in Dickie Lake based on the oxic model and the natural input, was much less than the measured concentration (6.7 vs 13.4 mg m^{-3}). If all of the potential input from shoreline development was included, the predicted concentration (12.5 mg m^{-3}) was slightly less than that measured. In the less likely case that the anoxic model is more suitable for Dickie Lake, the predicted and measured concentrations can be compared

Table 63. Areal water load (q_s; m yr^{-1}), TP retention coefficient (R_p; from equation 33), measured natural TP load (L_N; mg m^{-2} yr^{-1}), maximum possible TP load from shoreline development (L_A; mg m^{-2} yr^{-1}), maximum total load ($L_T = L_A + L_N$; mg m^{-2} yr^{-1}), measured TP concentration in the lake ($[TP]_{MEAS}$; mg m^{-3}), and predicted TP concentration based on natural load ($[TP]_N$) and total load ($[TP]_T$) using anoxic model (equations 27 and 33) for Chub and Blue Chalk Lakes.

	Chub	Blue Chalk
q_s	4.40	1.69
R_p	0.621	0.810
L_N	115.5	49.3
L_A	8.5	11.5
L_T	124.0	60.8
$[TP]_{MEAS}$	12.2	7.3
$[TP]_N$	10.4	5.8
$[TP]_T$	11.2	7.2

Table 64. Areal water load (q_s; m yr^{-1}), TP retention coefficient (R_p; from equation 32), measured natural TP load (L_N; mg m^{-2} yr^{-1}), maximum possible TP load from shoreline development (L_A; mg m^{-2} yr^{-1}), maximum total load (L_T; mg m^{-2} yr^{-1}), measured TP concentration in the lake ($[TP]_{MEAS}$; mg m^{-3}), and predicted TP concentration based on natural load ($[TP]_N$) and total load ($[TP]_T$) using oxic model (equations 27 and 32) for Harp and Dickie Lakes.

	Harp	Dickie
q_s	4.20	2.60
R_p	0.747	0.827 (0.735)
L_N	101.0	96.1
L_A	121.8	83.3
L_T	222.8	179.4
$[TP]_{MEAS}$	7.8	13.4
$[TP]_N$	6.4	6.7 (10.2)
$[TP]_T$	12.8	12.5 (19.1)

[1] The east basin of Red Chalk Lake was used for model calibration and therefore cannot be used to provide an error estimate.

in a similar manner. With no anthropogenic input, the prediction underestimates the measured concentration (10.2 vs 13.4 mg m^{-3}); while with 100% of the anthropogenic input, the prediction (19.1 of mg m^{-3}) overestimates the measured concentration. Calculation of the anthropogenic load from the observed concentration yields 29.4 mg TP m^{-2} yr^{-1}, which is 36% of the potential input.

To briefly recapitulate the principal findings of this section of the report:

i) A simple mass balance model to predict the TP concentration in a lake was described. This model was modified to suit conditions in the study lakes and may be described mathematically as follows:

$$[TP]^{1} = L\,(1 - R_p)\,/\,0.956\,q_s$$

ii) The model was tested on undeveloped Jerry and Red Chalk Lake (Main basin). The predicted and observed TP concentrations agreed to within 0.1 mg m^{-3}, giving confidence in the use of the model for developed lakes.

iii) The settling velocity (v) of TP in the lake was found to differ in lakes with oxic and anoxic hypolimnia. As the value of v affects the prediction of R_p in the model, this resulted in different model formulations for these two types of lakes. The value of v for a lake with an anoxic hypolimnion was set by calibration on the undeveloped east basin of Red Chalk Lake.

iv) Using the anoxic version of the mass balance model, it was demonstrated that 100% of the potential anthropogenic TP supply must have entered Chub and Blue Chalk Lake if the observed concentrations of TP were to be explained.

v) Similarly, using the oxic version of the model it was demonstrated that 100% of the potential anthropogenic TP supply must have entered Dickie Lake. However, only 19% of the potential anthropogenic TP supply need enter Harp Lake to explain the observed TP concentration.

The reasons for the difference between Harp Lake and Chub, Blue Chalk and Dickie Lake are perhaps twofold, and relate to the nature of the surficial deposits around the lakes and the age of the cottages on the lakes. The fraction of the potential anthropogenic TP supply from lakeshore development that enters the lake can be expected to vary with the type of sewage system used, the degree to which the system is functioning properly and the available TP binding capacity of the soil between the sewage system and the lake. The latter relates to the type of soil or surficial deposit around the lake. While no detailed information specific to the ability of soils to bind or retain TP was collected in this study, Jeffries and Snyder (1983) reported on differences in surficial geology between the study lakes. The surficial geology of Chub and Dickie Lake is primarily composed of thin till and rock ridges (61% and 81% respectively) whereas minor till plains dominate in Blue Chalk and Harp (77% and 50% respectively). Relatively thick deposits of outwash (5%) and sand (4%) are found in the Blue Chalk and Harp basins respectively. The age of the cottages may be related to the proportion of the TP binding capacity of the soils that remains available, the cumulative anthropogenic TP from older cottages having saturated the available TP binding sites. The average age of cottages (Downing, pers. comm.) around Harp Lake is less (1967) than it is around Chub (1961), Blue Chalk (1955) or Dickie Lake (1962). Additionally, it might be expected that the sewage disposal facilities (chiefly septic field systems) are in a better state of repair at Harp Lake because of their age.

C. EVALUATION OF THE EFFECTS OF LAKESHORE DEVELOPMENT ON AQUATIC BIOTA

Human activities associated with shoreline development of lakes may increase rates of nutrient supply to lakes. As the productivity and structure of aquatic communities may be nutrient dependent, increases in nutrient supply may have significant consequences for aquatic biota.

A variety of human activities may have detrimental impacts on aquatic life. Toxicants such as herbicides or petroleum residues may be introduced to lakes and impair the growth or survival of aquatic organisms. Littoral zone inhabitants may be sensitive to lake level manipulations or to siltation accompanying various activities in the watershed. Substantial impacts may accompany direct exploitation of particular communities such as harvesting of aquatic weeds or angling. Fishing may also indirectly influence benthic and plankton communities as removal of predators may change predation and grazing pressure on prey organisms.

The amount of development on the 15 study lakes was discussed in Section III.B.6. Human use ranged from 0-111 cap-yrs. yr^{-1}.$lake^{-1}$. Expressed areally and volumetrically, respectively, human usage ranged from 0 to 1.52 cap yrs ha^{-1} yr^{-1} and from 0 to 0.24 cap yrs.$10^4 m^{-3}$ yr^{-1} (Table 33). Trophic status of lakes is usually strongly correlated with phosphorus supply. The fraction of the total phosphorus supply potentially attributable to cottage domestic waste systems was calculated for six of the lakes. It ranged from 0% in Jerry Lake to 55% in Harp Lake (Table 45).

In this section, impacts of development on communities of phytoplankton, macrophytes and zooplankton are examined. Impacts are assessed by examining correlations between parameters describing the state of the biological communities and indices of lakeshore development or by comparing the values of the parameters in sets of lakes of extremely different intensities of use.

1. PHYTOPLANKTON

Pearson's product moment correlation coefficients were calculated between indices of human usage of the study lakes expressed in total, volumetric and areal units and several characteristics of the ice-free period phytoplankton communities of the lakes (Table 65). Several of these characteristics are known to be highly correlated with lake water TP concentrations (Nicholls and Dillon 1978, Dillon et al. 1978, Nicholls et al. 1983) and are therefore considered to be excellent indicators of the trophic status of the lakes.

Of 18 correlation coefficients calculated, the only significant ($p < 0.05$) correlation was between chlorophyll *a* and volumetric use. However, only 28% of the variance in chlorophyll *a* data was explained by its relationship with volumetric use.

Table 65. Correlation coefficients (r) between six characteristics of the phytoplankton communities of the study lakes and measurements of human use. Phytoplankton data were input as averages of the ice-free seasons of 1976-1979.

Parameter	Use Total	Use Areal	Use Volumetric
Total Biomass	0.36	0.32	0.44
Chlorophyll *a*	0.43	0.36	0.53
Maximum "*Bloomer*" Biomass[3]	0.15	0.24	–0.04
No. of Chlorophycean Genera	0.34	0.31	0.17
Species Richness[1]	0.39	0.33	0.24
Concentration of Dominance[2]	–0.28	–0.30	–0.18

[1] calculated as S-1/ln N; (Margaleff 1951), where S is the number of species (taxa) per sample and N the total number of individuals.

[2] calculated as $\Sigma (P_i)^2$ (Simpson 1949), where P_i is the proportion of the total sample belonging to the i^{th} species.

[3] maximum biomass of bloom-forming blue-green algae.

Development effects were then examined (t-test) by comparison of the means of several phytoplankton community parameters of two groups of lakes, the three most highly developed lakes and the four completely undeveloped lakes (Table 66). There were no significant differences ($p > 0.1$) between the two groups for the six phytoplankton community parameters. Clearly, factors other than human use alone are responsible for shaping the phytoplankton communities of these lakes.

2. MACROPHYTES

To assess whether development affected macrophyte community composition or nutritional status, macrophyte communities were surveyed in eight of the study lakes that reflected the full range of human use. Standing stocks of macrophytes were determined in Red Chalk and Harp Lakes, an undeveloped and a heavily developed lake, respectively.

Table 66. Four-year means of several phytoplankton community characteristics in three lakes with the highest estimated human use (developed) and the four undeveloped lakes. The t value for the difference between the means of groups and probability value (p) for rejecting a hypothesis of equal means of groups are indicated.

Lakes	Total Phytoplankton Biomass ($mm^3\ L^{-1}$)	Total Chlorophyll ($mg\ m^{-3}$)	Maximum Blue-green "Bloomer" Biomass ($mm^3\ L^{-1}$)	Number of Chlorophycean Genera	Species[1] Richness	Concentration[1] of Dominance
DEVELOPED						
Harp	0.66	2.75	0.94	32	80.6	0.09
Dickie	1.07	5.31	0.007	25	71.4	0.14
Walker	0.45	2.27	0.13	17	57.4	0.12
mean (s.d.)	0.73 (0.32)	3.4 (1.6)	0.36 (0.51)	24.7 (7.5)	69.8 (11.7)	0.12 (0.03)
UNDEVELOPED						
Jerry	0.79	3.09	0.10	25	65.4	0.20
Buck	0.52	2.27	0.005	18	54.4	0.14
Little Clear	0.45	2.15	0.23	20	56.9	0.15
Gullfeather	0.99	4.56	0.008	24	63.6	0.11
mean (s.d.)	0.69 (0.25)	3.0 (1.1)	0.08 (0.10)	21.8 (3.3)	60.1 (5.3)	0.15 (0.04)
t	0.18	0.41	1.08	0.71	1.51	1.32
p	0.80	0.60	0.20	0.50	0.10	0.20

[1] Defined in Table 48.

Table 67. Species richness, tissue concentrations in *E. septangulare* and macrophyte community biomass (with 95% confidence interval) in lakes. TP is the mean of 4 annual means (June 1976 – May 1980). Annual TP concentration and areal use of lakes as indicated.

Lake	Areal Use (M-yr $ha^{-1}\ yr^{-1}$)	TP ($mg\ m^{-3}$)	Richness (number of species $lake^{-1}$)	Mean Tissue Concentration[1] ($mg\ (g\ d.w.)^{-1}$) TP Plant	TP Sediment	TN Plant	TN Sediment	Total Biomass ($g\ m^{-2}$)
Chub	0.13	12.2	18	0.64	0.32	18.5	0.73	–
Crosson	0.085	11.7	19	1.00	0.29	19	1.10	–
Dickie	1.19	12.8	19	0.73	0.27	16	1.70	–
Gullfeather	0	13.4	19	0.74	0.30	17	0.55	–
Harp	1.52	7.8	18	0.47	0.35	14.2	0.67	73.6 ± 16.0
Little Clear	0	11.2	16	0.59	0.25	19	2.10	–
Red Chalk	0.033	5.7	13	0.69	0.56	14.3	0.51	59.2 ± 7.2
Solitaire	0.13	6.4	6	0.67	0.12	20	0.50	–

[1] Correlation coefficients between plant TN and sediment TN, plant TP and sediment TP, plant TP and water TP are 0.22 ($p < 0.10$), 0.06 ($p < 0.10$) and 0.35 ($p < 0.10$), respectively.

Divers recorded 33 taxa of submersed and floating-leaved vascular macrophytes, macroalgae (charophytes) and bryophytes in the eight lakes. All communities were dominated by "isoetid" or rosette species, particularly by *Eriocaulon septangulare* and *Lobelia dortmanna*, as is typical of softwater, oligotrophic lakes (Moeller 1975, 1978, Moyle 1945). Species richness was not correlated with areal human use (Table 67).

Vegetation cover was typically patchy and densest stands were largely confined to areas adjacent to inflowing streams. A distinct vertical zonation of species occurred in all lakes irrespective of development. Isoetids and floating-leaved species (*Nuphar*, *Nymphaea*, *Brasenia*) colonized water 3m deep. Bryophytes and *Nitella* sp. colonized deeper water.

Gerloff and Krombhotz (1966) have indicated that the growth of vascular macrophytes will be limited by phosphorus and nitrogen when tissue concentrations are <1.3 mg P g^{-1} and <13 mg N g^{-1}, respectively. Tissue concentrations of N and P in *E. septangulare*, the dominant species in all lakes are indicated in Table 67. Plant levels were not correlated ($p > 0.05$) with water or sediment nutrient levels and did not increase in developed lakes. Assuming applicability of the Gerloff and Krombhotz criteria, *E. septangulare* was P-limited in all lakes regardless of development. Tissue levels of N indicated that supplies of this nutrient were sufficient.

Given that development had not increased nutrient availability, it is perhaps not surprising that total macrophyte community biomass was similar in Harp Lake, a heavily developed lake (73.6 ± 16 g m^{-2}) and in the relatively undeveloped Red Chalk Lake (59.2 ± 7.2 g m^{-2}, Table 67).

These comparisons indicate indirect effects of shoreline development operating through changes in nutrient supply have no detectable significant effects on the composition, standing stocks or nutritional state of aquatic macrophyte communities of the lakes. Other direct interventions related to development, such as beach clearing, would, of course, have negative impacts on vascular macrophytes.

3. ZOOPLANKTON

Averages for 1977 to 1979 for ten zooplankton community parameters were assembled for the 15 study lakes (Table 68). To test whether development influenced zooplankton community structure or standing stocks, t tests of difference between means were performed on two groups of lakes, the three most highly developed lakes (Harp, Dickie and Walker) vs. the four completely undeveloped lakes (Jerry, Buck, Little Clear and Gullfeather).

Development might have been expected to increase nutrient supply, yet neither total nor herbivorous zooplankton biomass, nor the biomass of Calanoida, Cyclopoida or Cladocera were significantly higher ($p > 0.05$) in the developed lakes. Development similarly did not influence the average richness of the zooplankton communities. Richness averaged 9-10 species per collection in each group of lakes.

The proportional biomass of herbivores vs. carnivores or of cladocerans and copepods were not different in the two lake groups. Finally, the zooplankton were of similar average size in the developed and undeveloped lakes, indicating that changes in zooplankton predators or prey produced by development, did not have a large impact on zooplankton size structure.

The range in zooplankton parameters was generally greater among the undeveloped than the developed lakes (Table 68). This variability is clearly produced by some factor other than human usage of the lakes.

Table 68. Comparison of various zooplankton parameter means between three developed study lakes (Harp, Dickie, Walker) and 4 undeveloped lakes (Jerry, Buck, Little Clear, Gullfeather). t value and probability value for rejecting the hypothesis of equal means are indicated.

	Areal Use	Crustacean Zooplankton Biomass (mg m^{-3})					Percent Biomass			Richness	Size
Lake Group	(m-yr ha^{-1} yr^{-1})	Total	Herbivore	Calanoida	Cyclopoida	Cladocera	Herbivore	Cladocera	Copepoda	(No. of species/ collection)	(µg/ animal)
Developed											
mean	1.21	47.4	37.6	12.6	9.4	23.9	79	50	50	9.8	0.73
lowest value	0.93	40.2	30.1	11.8	8.0	19.8	75	49	47	9.1	0.66
highest value	1.5	53.5	43.2	13.5	11.8	25.7	81	53	51	11.0	0.85
Undeveloped											
mean	0	61.6	45.9	5.2	14.8	40.3	74	61	39	9.1	0.66
lowest value		32.5	25.8	1.5	7.0	16.8	62	35	24	8.2	0.54
highest value		95.1	74.1	13.7	19.9	72.7	79	76	65	10.4	0.71
t	8.49	0.86	0.6	2.14	1.59	1.04	1.05	1.01	1.01	0.93	1.11
p	0.001	0.50	0.60	0.10	0.20	0.40	0.40	0.40	0.40	0.40	0.40

IV. DEVELOPMENT AND CALIBRATION OF MODELS FOR PREDICTING THE EFFECTS OF DEVELOPMENT ON LAKE TROPHIC STATUS

In Part III of this report, a combination of existing and newly formulated models were used to evaluate the contribution of lakeshore development to the TP content of the "A" lakes. In this section, generalized models are developed and calibrated for predicting the effects of shoreline development on the trophic status of soft-water lakes in the Precambrian regions of Ontario.

The variables that have traditionally been used to describe the trophic status of lakes include the chlorophyll *a* concentration (a measure of phytoplankton abundance), phytoplankton biomass, phytoplankton species composition (especially the presence or absence of indicator species, e.g., bloom-forming blue-green algae), water clarity (usually measured as the Secchi disc depth), and areal hypolimnetic oxygen depletion rate (AHOD). These variables have been focused on for several reasons: a)they are "state variables"; that is, they integrate many components of the lake system, b)they are relatively easily measured, and c)they represent measures of trophic status useful to the layman since they are readily visible or have readily visible effects (e.g., high AHOD may result in fish kill).

The conceptual model that linked these state variables to the water and TP mass balances of lakes was outlined in Figure 1. In the following sections, models suitable for predicting water and TP balances are formulated, and linkages between these models and state variables defining trophic status are presented.

The possibility that acidic deposition has altered the relationships linking trophic status parameters is also considered in this section. The chlorophyll *a*, phytoplankton biomass and water clarity models are derived with consideration of both nutrient enrichment and acidification parameters, as is the zooplankton biomass model discussed in Section V.

A. HYDROLOGIC MODELS

All important components of the hydrologic regimes of the "A" lakes were directly measured in this study as this information was needed to determine the natural inputs of total phosphorus to the lakes. In fact, one criterion for selecting these study lakes was that collection of such data was technically feasible. It is costly, time consuming and may not always be technically feasible to assemble measured hydrological data for all lakes on which development may occur. It would therefore be of great aid to lake managers if models which accurately predict important components of the hydrologic cycle of Precambrian Shield lakes could be developed. The possibility of constructing such models is investigated in this section.

Groundwater flux and changes in lake storage were quantitatively unimportant for the six "A" lakes (Section III.B.2) as previously observed for other Precambrian lakes (Schindler et al. 1976, Newbury and Beaty 1977, Scheider 1983). Precipitation to the lake surface and terrestrial runoff were the major sources of water to the lakes. Loss via the lake outlet and evaporation were the major losses. If these four terms of the water balance equation for lakes could be accurately predicted, then the hydrologic regime of most lakes would be well characterized.

For a terrestrial watershed, precipitation, runoff and evapotranspiration are the three important components in the annual water balance. As the time interval of interest decreases (years to months to weeks), changes in storage (either in lakes or in soil) become more important in the water balance. Given the primary interest in using the data to predict average water quality changes, annual timesteps are generally adequate. For annual timesteps, changes in storage may be ignored.

1. MODELLING PRECIPITATION DEPTH

The simplest method of predicting precipitation depth at a given location is to use the measured value for that time at a nearby meteorological station. Many such stations operate continuously in Ontario (Hydrological Atlas of Canada 1978). Precipitation depth was relatively uniform between the seven stations in the study area, (i.e., no significant differences in depth among stations on an annual basis or on a monthly basis in 10 of 12 months), indicating that annual and monthly precipitation depth at a given lake in the study area may be estimated using data from the closest meteorological station. To improve estimates for shorter time durations (daily) or across areas for which precipitation depth does vary, a weighted average of the two closest meteorological stations (Viessman et al. 1977) may be calculated as follows:

$$P_{i1} = \Sigma_j P_{ij} \cdot (W_j / \Sigma_j W_j) \qquad (34)$$

where: P_{i1} = precipitation for day i derived for a given location number 1

P_{ij} = precipitation for day i at location j

$W_j = 1/D_j^2$

D_j = distance from given location to meteorological station j.

Other methods of obtaining precipitation depth using data from several surrounding stations are the Thiessen polygon method and the isohyetal method (outlined in Gray 1970).

No reliable method exists for predicting precipitation in the future; however, a range of possible values at a particular location can be generated using long-term records such as those summarized in the Hydrological Atlas of Canada (1978).

2. MODELLING RUNOFF AND EVAPOTRANSPIRATION

Runoff from the terrestrial basin supplied the largest quantity of water to the study lakes. Loss via the outlet was the major loss (Tables A1-A6). There are many approaches to predicting surface runoff (summarized in McMahon 1979). The strengths and weaknesses of five techniques are summarized below, along with their required data inputs.

a) Annual runoff may be assumed to equal long-term average runoff. Coulsen (1967), Pentland (1968) and Hydrological Atlas of Canada (1978) present isolines of average runoff drawn from information at many gauged rivers and streams in Ontario. Errors in estimates generated using this method can arise due to local variability in precipitation depth, evapotranspiration rates and catchment characteristics such as physiography, forest cover and soil type. Of course, as the data are long-term average values, errors in prediction will occur in any given year because of climate variability. Pentland (1968) also gives long-term mean runoff isolines for Ontario on a monthly basis. However, one can expect progressively greater divergence between predicted and actual runoff the shorter the time interval for prediction. Year-to-year variability in monthly runoff in the 1976-1980 period was substantial for the "A" lakes (Table 69), particularly in October and March.

b) A second method for predicting annual runoff from ungauged watersheds in Muskoka-Haliburton from 1976-80 is to use the mean runoff of the gauged "A" lake and export watersheds measured in the same year. This overcomes problems associated with the use of long-term mean runoff data. The fractional distribution of annual runoff between months differed in the headwater watersheds compared to the lake outlets (Table 69). Predictions for ungauged watersheds should therefore employ appropriate proportions of annual runoff.

c) The second method corrects predictions of runoff only for year-to-year differences in some meteorological characteristics, but does not account for differences among watersheds. Multiple linear regression was used as a technique to attribute the variability in streamflow characteristics among watersheds to various watershed characteristics. As expected, total annual discharge was highly related to watershed area in each of the four years ($r^2 \simeq 0.98$). The slope of the regression line between watershed area and annual discharge is in effect the mean runoff, so predicting annual streamflow in this manner yields similar (differing by the value of the intercept) results to those predicted by the second method.

Four watershed characteristics explained 45% of the variance in mean (1976-1980) annual runoff ($r^2 = 0.448$; $n = 36$). Mean annual runoff (R, mm yr^{-1}) was a function of stream length (L, m), % of the watershed area underlain by organic deposits (OR, %), % of the watershed area underlain by shallow till (ST, %), and stream length/-watershed area (L/A, m^{-1}) as follows:

$$R = 355 + 0.043\,(L) + 6.29\,(OR) - 0.91\,(ST) + 1.98\,(L/A)\,10^4$$

The above three techniques use measured runoff values at gauged watersheds to estimate runoff at ungauged watersheds. The estimates use either long-term mean values or apply only to the 1976-1980 period. More general models are necessary to allow prediction of runoff for any given set of climatic conditions or to predict runoff for watersheds with different characteristics than those of the "A" lakes.

d) If change in storage and groundwater flux are assumed to be zero (a reasonable assumption on an annual timestep), then runoff will equal the difference between precipitation and actual evapotranspiration. Hence annual runoff may be predicted from measured annual precipitation if independent predictions of evapotranspiration are available.

Evapotranspiration may be estimated in four ways. The mass balance method keeps an account of the components of the water balance equation and arrives at evapotranspiration by difference. The energy balance method requires measurement of components of an

Table 69. Mean % annual stream discharge occurring in each month for 23 headwater streams and 6 lake outlets for the 1976-1980 period.

	23 Headwater Streams			
Month	1976-77	1977-78	1978-79	1979-80
June	1.65	0.77	2.18	2.40
July	3.19	0.58	0.63	0.94
Aug.	0.95	0.77	2.26	1.47
Sept.	0.80	2.72	2.58	1.19
Oct.	1.71	10.03	6.48	8.33
Nov.	4.71	14.51	8.91	16.39
Dec.	5.02	9.20	6.36	10.18
Jan.	2.17	3.95	3.50	7.31
Feb.	1.50	1.75	2.13	1.58
Mar.	39.82	2.95	26.56	10.20
Apr.	34.22	39.75	28.57	34.81
May	4.23	12.91	9.82	5.14
	6 Lake Outlets			
Month	1976-77	1977-78	1978-79	1979-80
June	2.51	0.47	3.20	3.61
July	1.97	0.34	0.50	0.39
Aug.	0.69	0.46	1.33	1.05
Sept.	0.31	0.71	2.15	1.40
Oct.	0.47	8.34	4.41	6.19
Nov.	2.74	15.09	8.06	15.47
Dec.	6.97	12.85	8.71	13.82
Jan.	3.72	7.20	7.11	9.57
Feb.	2.89	3.46	3.61	3.44
Mar.	29.06	3.88	21.65	8.05
Apr.	43.47	29.69	27.08	31.37
May	5.11	17.48	12.18	5.67

energy balance equation to calculate evapotranspiration. The vapour transfer method considers the physical process of evaporation, i.e., evaporation is proportional to the vapour pressure gradient at the surface. Other indirect methods make use of meteorological information available at most climatic stations. The first three methods require many forms of instrumentation to obtain values of necessary input parameters. As this instrumentation is generally not in place, these methods are unsatisfactory for predicting evapotranspiration for most watersheds.

Several indirect methods are available. Penman (1948) developed a semi-empirical formula to predict free water evaporation which he multiplied by a constant to obtain actual evapotranspiration. Turc (1954, 1955) developed formulae to relate annual actual evapotranspiration to precipitation. Application of Turc's formula for shorter time periods (<annual) requires a crop factor. Finally, Morton (1965, 1968, 1969, 1971, 1976, 1978) and Morton et al. (1980) developed an approach to predict actual evapotranspiration from information on air temperature, dew point temperature and the ratio of observed to maximum possible sunshine duration or the observed global radiation. The simplicity and availability of the data required to run the model and the good agreement with long-term water balance estimates of areal evapotranspiration (Morton 1978) made this the most attractive model.

Using Morton's method, estimates of evapotranspiration were calculated for the Harp and Jerry Lake watersheds. These estimates were compared with direct measurements (a weighted average water loss over all watersheds) derived from the measured water balance. Discrepancies were observed on a monthly basis (Figure 13) but this was expected because changes in storage are not considered by the water balance estimate. However, over the 1976-1980 period, the total difference in evapotranspiration estimates was only 31 mm (1957 mm estimated by the water balance method vs 1988 mm estimated by Morton) or <2% (Figure 14). The good agreement between the two methods gives confidence in the use of Morton's method to estimate evapotranspiration and hence runoff on an annual basis. However, for shorter time steps (e.g., monthly), changes in storage must also be taken into account if runoff is to be predicted from precipitation and evapotranspiration.

Because the input data to Morton's model were the same for all Harp and Jerry Lake watersheds, the model predicted that all watersheds would have the same evapotranspiration and, because the precipitation depth is also the same at the two lakes, the same runoff. The observed range in mean (1976-1980) annual runoff (0.29-0.56 m yr^{-1}) among the watersheds of Harp and Jerry Lake, presumably due to differences between watershed characteristics, cannot be addressed using Morton's model.

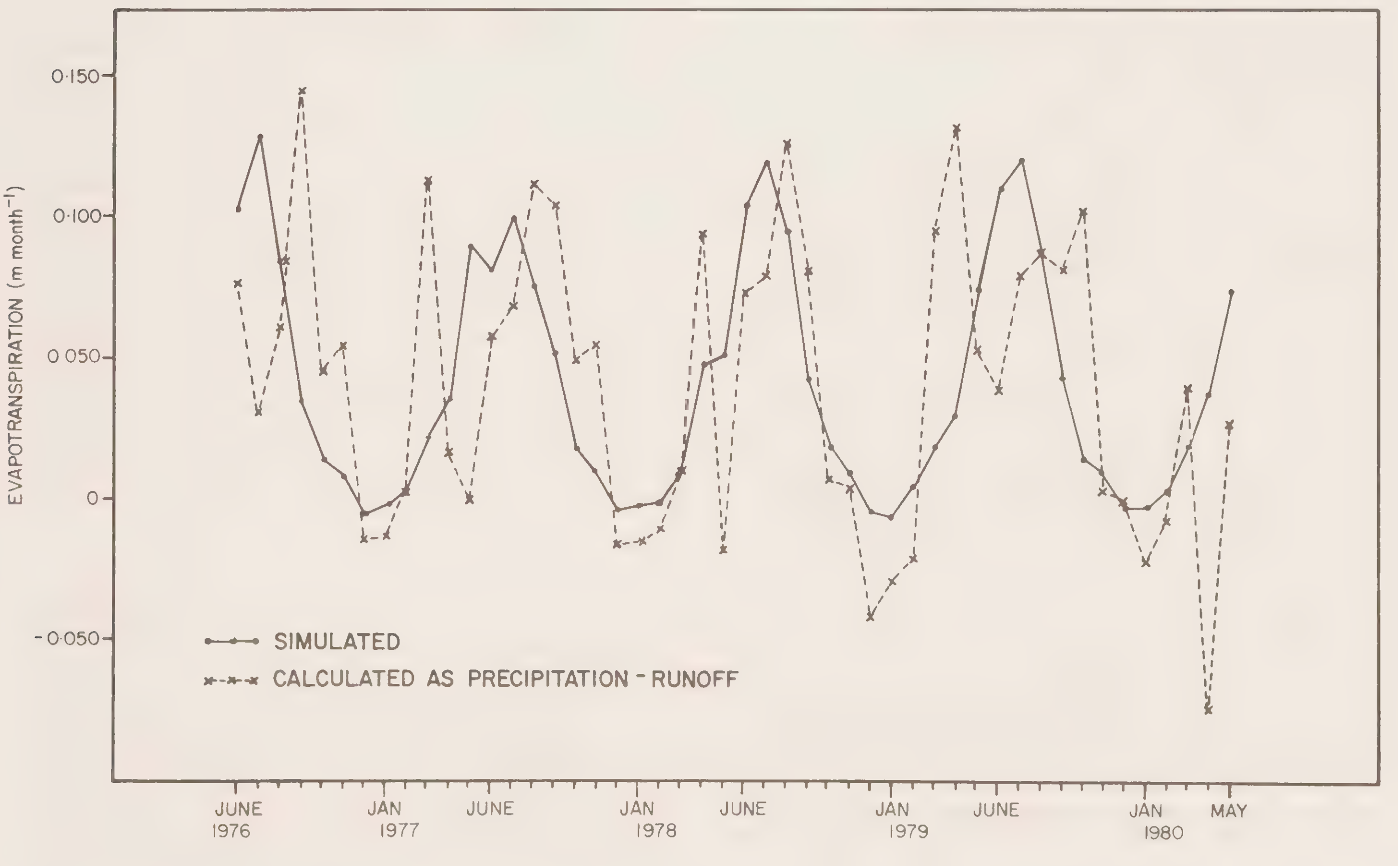


Figure 13 Monthly evapotranspiration losses estimated using the method of Morton et al. (1980) (solid line) compared to losses calculated as the difference between observed precipitation and runoff (dashed line) for watersheds of Harp and Jerry Lake (1976-1980).

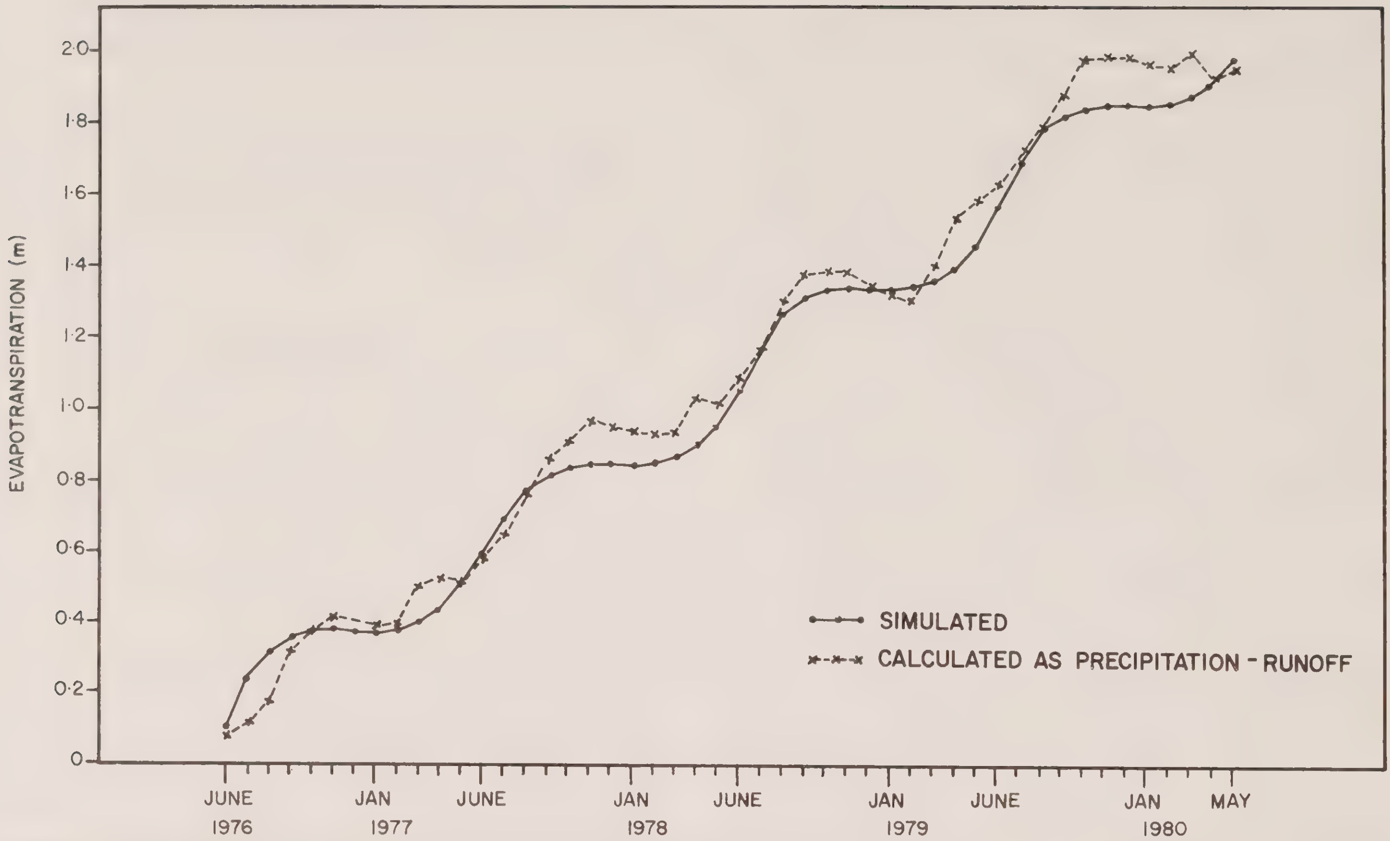


Figure 14 Cumulative plot (1976-1980) of monthly evapotranspiration losses estimated after Morton et al. (1980) (solid line) and losses estimated as the difference between observed precipitation and runoff (dashed line) for watersheds of Harp and Jerry Lake.

e) Finally, runoff is the output of a range of process models (many are summarized in Fleming 1975) which simulate the movement of water through the drainage basin. These models predict flow on a very fine time scale (daily) for any given set of climatic conditions. Several such models have been used in chemical mass balance studies (e.g., Christopherson et al. 1982, Chen et al. 1982). A model developed by the Tennessee Valley Authority (1972) was adapted for use in Precambrian areas of Ontario (Goebel and Thysen 1982, Goebel 1982, 1983). This model was chosen because it provides daily flow estimates using readily obtainable meteorological data as inputs (Logan 1980).

The Tennessee Valley Authority (TVA) model is based on the concept of water balance (Figure 15). All incoming moisture is allocated by the model such that precipitation – evapotranspiration – streamflow – Δ storage = 0. The model operates in a cascading fashion to allocate precipitation to the various compartments shown in Figure 15. The ultimate fate of incoming precipitation is loss to evapotranspiration or streamflow. Deep seepage losses are set to zero for the study watersheds.

The model considers precipitation to be rain if the mean daily air temperature ≥1.5°C. Rain and snow are first lost to interception then cascade to the next compartment. The quantity of accumulated snow which melts in a day is a function of mean daily air temperature and rainfall as follows:

$$\text{melt (mm)} = 1.3 \text{ (air temp. °C) (if air temp.} > 1.5\text{°C)} \quad (35)$$

$$\text{melt (mm)} = (3.5 + 0.012 \times \text{precip.}) \text{ air temp. °C} + 1.2 \text{ (if air temp.} > 1.5\text{°C, if rain)} \quad (36)$$

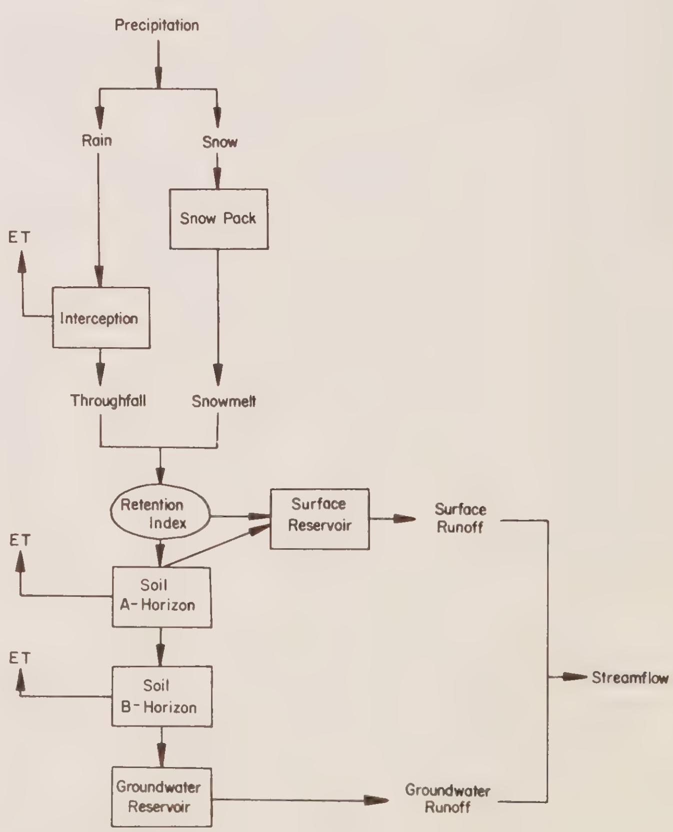


Figure 15 Schematic outline of the modified TVA streamflow model.

The snowpack itself is assumed to have a 3% water retention capacity. This part of the model shifts the time when precipitation input is available for runoff routing.

In either case, (rain or snow), the incoming precipitation, after losses to interception and storage or release by the snowpack, is potentially available for surface runoff. The amount allocated to surface runoff is between 0 and 100%, depending upon the retention index. If the soil is saturated with moisture, the retention index assumes a value of 0 and all incoming precipitation is available for runoff. Conversely, under dry soil conditions the value of the retention index approaches 1 and little water is available for surface runoff.

The remainder of the water is potentially available for groundwater runoff (baseflow). It first cascades into the A soil compartment (maximum storage capacity = 25 mm). If this compartment is saturated, the remaining water is routed to surface runoff. Water from the A soil compartment is transferred to the B soil compartment and ultimately into the groundwater reservoir. Losses from the groundwater reservoir are calculated from a groundwater recession curve. Groundwater plus surface runoff combine to yield total streamflow. The model contains several routing equations which determine the rate of surface and groundwater runoff.

In addition to streamflow, water is lost by evapotranspiration (Morton et al. 1980). Evapotranspiration demands are first met by the water stored in the interception compartment, then by water from the A soil compartment and finally by water from the B soil compartment.

The data required to utilize the model are daily precipitation, mean daily temperature and the data needed to compute evapotranspiration after Morton et al. (1980). In addition, the model requires that values for a number of parameters be fixed. These include such parameters as the beginning of the spring, summer, fall and winter seasons, maximum moisture storage by the interception, A and B soil compartments etc. There are nine parameters which are optimized in the model. An optimization routine which sets the value of these nine parameters by minimizing the sum of squared differences between the observed and predicted streamflow is included in the model calibration procedure. The model runs on a time period of October – September. Prior to this, there is a 3-month lead in period to assist in setting values of various model parameters.

The modified TVA model was calibrated using data for the period Oct. 1, 1976 to Sept. 30, 1979 for the 21 gauged inlets to the "A" lakes. Overall correlation coefficients for the 3 year period ranged from 0.57 at Harp inlet 3 to 0.76 at Chub inlet 1. The predicted and observed daily runoff for Harp inlet 4 are shown in Figure 16 (a,b,c) as a sample result. Major fall storm events were correctly simulated by the model as were low runoff values during the winter and summer periods. Runoff during the spring period was modelled quite well for 1977 and 1979 but the simulated runoff did not agree with the observed runoff in the spring of 1978. The reason may be that in 1978 the spring runoff occurred relatively late. In the TVA model, values of various parameters vary seasonally but the beginning of the seasons is fixed. A varying season parameter would probably improve the model results.

The model was tested in two ways: by extending the runoff records in time for watersheds on which the model had been calibrated and by predicting runoff at watersheds for which the model had not been calibrated. In the former test, the model was used to predict runoff for the period Oct. 1, 1979 to May 31, 1980 for the 21 gauged inlets of the "A" lakes. The predicted values of runoff were compared to observed data. Values of the correlation coefficient between predicted and observed runoff ranged from 0.55 to 0.89. A sample result (Harp inlet 4) is shown in Figure 16 (d).

In the second test, there were no model calibration runs against observed runoff data to set values for the nine model parameters and these had to be estimated by other means. A multiple linear regression analysis using 21 estimates of each of the nine model parameters (data obtained from model calibration runs on the 21 gauged inlets to the "A" lakes) was carried out using measures of the geological and physical characteristics of the water-

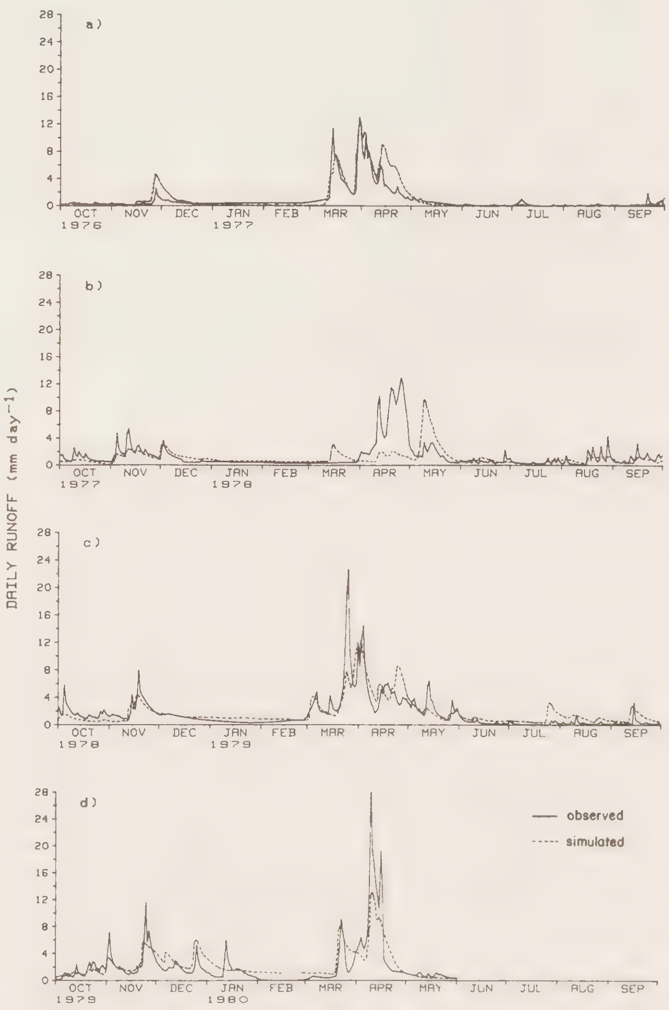


Figure 16 Comparison of simulated (TVA model, dashed line) and observed (solid line) daily runoff (mm day^{-1}) of Harp inlet 4 for the period during which the model was calibrated (Oct. 1976 – Sept. 1979) and for a test period (Oct. 1979 – May 1980).

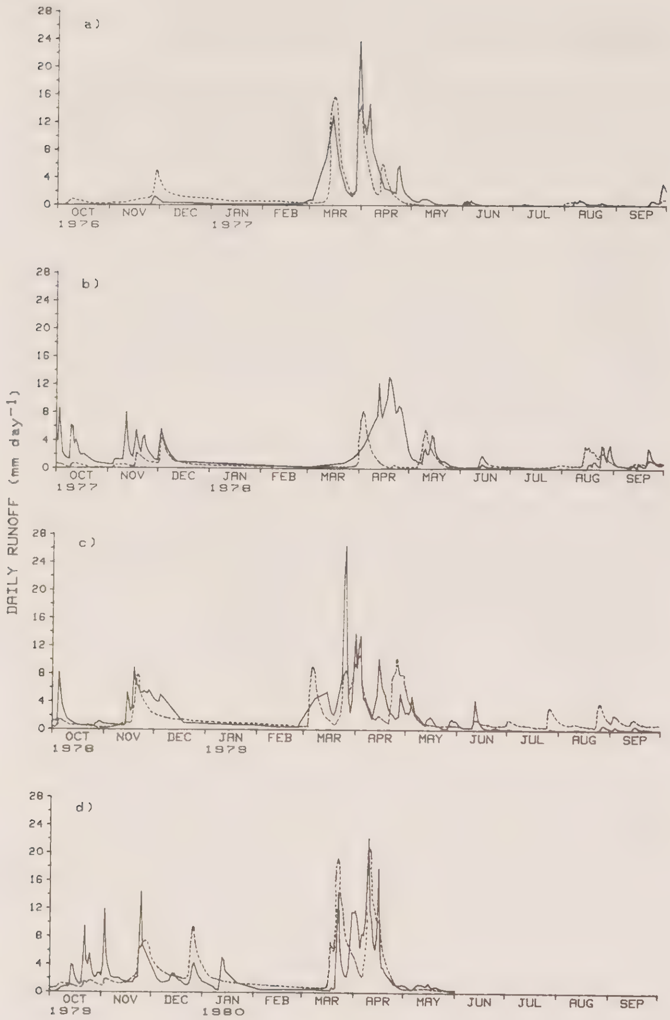


Figure 17 Comparison of simulated (TVA model, dashed line) and observed (solid line) daily runoff (mm day^{-1}) for Paint inlet 1 for the test period Oct. 1976 – May 1980.

sheds as independent variables. Although the results of all the regression analyses are not reported here, it was found that four independent variables (watershed gradient, and the % watershed area covered by woodland, organic deposits and exposed bedrock) explained 42-92% of the variance in some model parameters. The mean value of the parameter was used where no relationship with the independent variables was found. Using these relationships, the TVA model was tested against observed runoff data for eight of the nine "export" watersheds for the period Oct. 1, 1976 – May 31, 1980. Values of the correlation coefficient between predicted and observed runoff ranged from 0.05 to 0.91. As was observed in the calibration runs on the 21 gauged "A" lake inlets, the poorest agreement between the predicted and observed runoff generally occurred in the spring of 1978. Results for Paint inlet 1 are shown in Figure 17 as a sample.

3. MODELLING EVAPORATION

As with precipitation and runoff, lake evaporation may be predicted using long-term mean annual records (Hydrological Atlas of Canada 1978). Of course, year-to-year variations are not addressed by this method. The evaporation isopleths are based on modified pan evaporation data, and are available from certain climatic stations on an annual as well as mean long-term annual basis.

Lake evaporation during 1976-80 for lakes in Muskoka-Haliburton may also be predicted using the mean annual evaporation measured for the "A" lakes (Table 12). This technique is acceptable for prediction on an annual basis because no significant ($p > 0.05$) differences occurred in lake evaporation between the seven lakes. However, it is less justifiable on a monthly basis because significant ($p < 0.05$) differences in evaporation between lakes occurred in some months. This method of predicting evaporation is less certain on a broader spatial scale.

A more general method of predicting lake evaporation based on climatological data was presented by Morton (1979) and Morton et al. (1980). The data needed are the same as those for his evapotranspiration model. Using Harp Lake as an example, evaporation estimated by the Morton method and by the energy balance method showed similar seasonal patterns (Figure 18), although Morton's method estimated higher evaporation rates during the early part of the open water period (April – July) and lower evaporation rates during the later period (August – December). The annual evaporation estimates agreed closely, Morton's method being 2.5%-7.6% higher, in part because the energy balance method assumed zero evaporation during the period of ice cover.

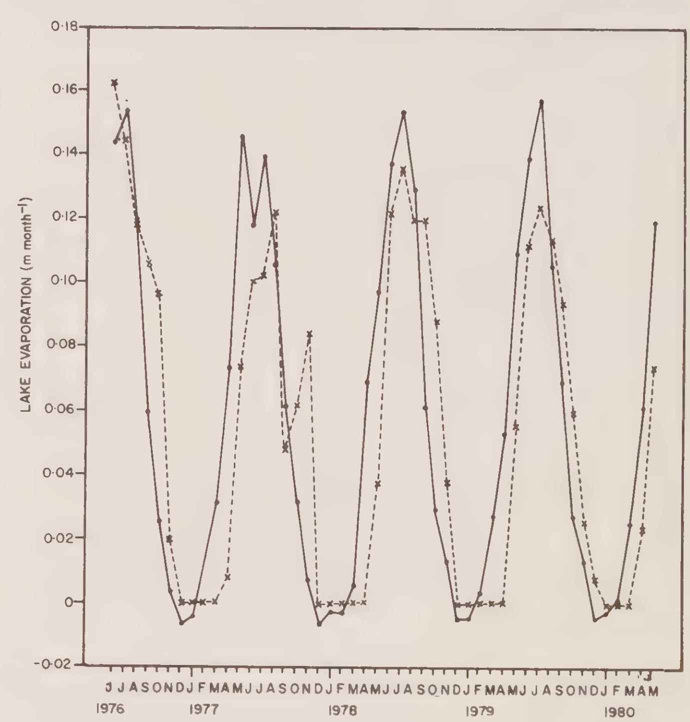


Figure 18 Comparison of monthly lake evaporation estimated by the energy balance method (dashed line) and by the method of Morton et al. (1980) (solid line) for Harp Lake (1976-1980).

B. MODELLING THE EXPORT OF TOTAL PHOSPHORUS

The flux of total phosphorus was measured from all major watersheds of the "A" study lakes. However, because of time, cost or technical limitations it may not always be feasible to obtain these measurements for other lakes.

Therefore, models to predict TP export from readily obtainable information would be of assistance to lake managers. Modelling the TP export from watersheds is discussed in this section with specific reference to developing models for application to the "B" lakes. Analysis of the TP export data for the 21 gauged "A" lake watersheds and nine "export" watersheds (Section III.B.5) demonstrated that spatial variability in TP export was greater than year-to-year variability. Emphasis for modelling the TP export was placed accordingly.

Earlier attempts to relate TP export to watershed characteristics (Vollenweider 1968, Dillon and Kirchner 1975a, Kirchner 1975) have been discussed previously. The new models attempted to refine these relationships, primarily for watersheds classified by Dillon and Kirchner as "forested, igneous", by considering an expanded set of watershed characteristics. The stepwise strategy for multiple linear regression was used to derive the models (Draper and Smith 1981). Forty-six variables representing hydrologic, geologic and land use characteristics were employed as independent variables and TP export from 21 "A" lake watersheds and nine additional "export" watersheds as dependent variables. Data on land use and geology were assumed constant over the four-year period (1976-1980). The annual TP export data were summarized in Tables 22 and 23. Table 70 lists the 46 variables describing watershed characteristics.

Table 70. Variables used in regression analysis on annual export of total phosphorus.

Type of Variable	Source of Data	Variable (V) Number	Definition of Variable
Hydrology	*	1.	Annual precipitation (mm)
	*	2.	Annual runoff (mm)
	++	3.	Yield (ratio) [=V2/V1]
	*	4.	Annual maximum mean daily discharge/mean annual mean daily discharge (ratio)
	*	5.	Annual base flow/total annual flow (ratio)
	*	6.	Spring flow [March + April + May total/total annual flow] (ratio)
Geology	**	7.	Area of watershed (m^2)
	+	8.	Grade (% slope)
	+	9.	Runoff coefficient
	**	10.	Length of gravel roads (km)
	+	11.	Stream length of main channel (m)
	++	12.	Area/stream length (m^2/m)
Bedrock geology	**	13.	Biotite gneiss (% total area)
	**	14.	Amphibolite and schist (% total area)
	**	15.	Migmatite (% total area)
	**	16.	Marble (% total area)
	(a) **	17.	Meta-arkose and minor marble (% total area)
	**	18.	Biotite gneiss and arkose marble (% total area)
	(a) **	19.	Diorite (% total area)
	++	20.	Carbonate bedrock (% total area) [V16 + V17 + V18]
	++	21.	Silicate bedrock (% total area) [100 – V20]
	++	22.	Mafic bedrock (% total area) [V14]
	++	23.	Felsic bedrock (% total area) [V13 + V15 + V19]
Surficial geology	**	24.	Carbonate till (% area)
	**	25.	Minor till plain (% area)
	**	26.	Thin till and rock ridges (% area)
	**	27.	Peat (% area)
	**	28.	Bedrock (% area)
	**	29.	Outwash plain (% area)
	**	30.	Sand (% area)
	(a) **	31.	Esker (% area)
	(a) **	32.	Drumlin (% area)
	**	33.	Pond (% area)
	++	34.	Deep overburden (% area) [V25 + V29 + V30 + V31 + V32]
	++	35.	Shallow overburden (% area) [V26 + V28]
	++	36.	Organic overburden (% area) [V27 + V33]
Relative abundance	***	37.	Sources (0 = none, 1 = some, 2 = many)
of nutrient	***	38.	Sinks (0 = none, 1 = some, 2 = many)
sources and sinks	***	39.	Absence of sources or sinks (0 = presence, 1 = absence)
Land use	+	40.	Flat woodland (% area)
	+	41.	Rolling woodland (% area)
	+	42.	Hilly woodland (% area)
	+	43.	Pasture (% area)
	(a) +	44.	Cultivated (% area)
	+	45.	Water (% area)
	+	46.	Bog (% area)
	+	47.	Wetland (% area)
	(a) +	48.	Urban (% area)
	(a) +	49.	Flat bedrock (% area)
	+	50.	Rolling bedrock (% area)
	+	51.	Hilly bedrock (% area)
	+	52.	Total bedrock (% area)

Sources of Data

* * Ministry of the Environment: data obtained from periodic measurements and computed on an annual basis.
* ** Ministry of the Environment: data obtained from air photo interpretation, on-site validations, and collaboration with published geology maps.
* *** Ministry of the Environment: data obtained through inspection of air photos.
* \+ Ministry of Transportation and Communications: data obtained by airphoto interpretation.
* ++ Data computed from other variables.
* (a) Variable was eventually deleted from the analysis as it represented a zero vector or near-zero vector in the regression analysis.

Two tiers of models were constructed, one using all independent variables and a second excluding the hydrologic variables. Each tier of models included a regression model for each of four years (1976-77, 1977-78, 1978-79,

1979-80) as well as a model for the entire four-year period. No analysis of residuals was done. Missing data were deleted from the analysis on a listwise basis only.

Using TP export, hydrologic, geologic and land use data from all four years (1976-1980), the stepwise procedure (at the 90% confidence level) produced a final equation after five steps which accounted for 51% ($r^2 = 0.51$) of the variance in TP export. Annual TP export (mg m^{-2} yr^{-1}) was predicted as a function of peat (% area), runoff (mm yr^{-1}), grade (% slope), bedrock (% area), and pasture (% area). These results are given in Table 71, model A.

Using TP export, geological and land use data from all four years, a model was constructed (at the 90% confidence level) after four steps which accounted for 46% of the variance in TP export ($r^2 = 0.456$). As might be expected, the variables entered the equation in the same order as the previous model. The estimate of annual TP export (mg m^{-2} yr^{-1}) was predicted as a function of peat (% area), grade (% slope), bedrock (% area), and outwash plain (% area). The results are given in Table 71, model B.

Although the results of the regression analyses for data from individual years are not reported here in detail, it should be noted that these models were simpler (i.e., fewer predictor variables entering the regression equations) and less powerful (r^2 values <0.5) than the models constructed on all four years' data. The variables peat and organic overburden were important predictors of annual TP export in all the models, both the single-year models and the four-year composite models. They usually entered first in the stepwise regression procedure and in some cases were the only significant variables in the single-year models.

C. MODELLING THE TP CONCENTRATION IN LAKES

The trophic status of most Precambrian Shield lakes is governed by the concentration of TP in the lake, which in turn is a function of the lake's TP mass balance, hydrologic budget and morphometry (Figure 1). Models which link these parameters to predict changes in the TP concentration of a lake have been discussed in Section III.B.10 and only a brief review will be given here.

Early mass balance models to predict the chemical content of lakes were reviewed by Dillon (1974). A simple model to predict the TP concentration in a lake at steady-state was expressed (equation 16) in the following form:

$$[TP] = J_T / V\phi$$

Alternatively, if sedimentation loss of TP is expressed in terms of a sedimentation velocity (v), the model was expressed (equation 18) as:

$$[TP] = J_T / (Q + vA)$$

A third model formulation (equation 19) employed a retention coefficient, R_p, to describe the sedimentation of TP.

$$[TP] = J_T (1 - R_p) / Q$$
$$\text{or } [TP] = L (1 - R_p) / q_s$$

Many alterations have been proposed to this simple mass balance model and several improvements were incorporated in this study, following the approach that the best model is the simplest one that makes accurate predictions. Mass balance models which are semi-empirical, such as these, may best predict TP concentration on an annual basis or at least TP concentrations representing most of the year. Because one generally wishes to predict trophic status parameters during the ice-free period, the mass balance model was applied to predict mean ice-free concentrations of TP. To allow the ice-free period model to predict TP concentrations in other time periods, the ice-free TP concentrations were related (Table 48) to values of TP in other time periods (spring overturn, summer stratified, fall overturn, winter).

Differences in TP concentrations between vertical depth zones were observed in some lakes (Table 49). However, Snodgrass and Dillon (1983) compared one and two-layer mass balance models using a data set collected in Muskoka-Haliburton (Dillon 1974) and found that the one-layer model provided equally accurate predictions of TP concentration as did the two-layer model. Based on these results, the simple model described in equation 19 was not modified to account for thermal stratification.

Inherent in this simple model is the assumption that the concentration of TP in the lake and in the lake outlet are equal. Using data from the "A" lakes, the following modification was developed (equation 26).

$$[TP[_{OUT} = 0.956\ [TP]_{IF}$$

Employing the above model revisions, the simplest version of the TP mass balance model (equation 27) was formulated as:

$$[TP] = L (1 - R_p) / 0.956\ q_s$$

This version of the model was termed LCS-1. Alternatively, employing information on sedimentation loss of TP derived from whole-lake Pb burdens, Pb-210 accumula-

Table 71. Stepwise regression models for the prediction of gross annual export of total phosphorus (mg m^{-2} yr^{-1}). Independent variables and their units are listed in Table 70. All types of independent variables were used in model A whereas only geology and land use variables were used in model B.

Model	Step 1	Step 2	Step 3	Step 4	Step 5	Constant	SE	p
A[1]	0.38V27	0.012V2	–0.33V8	–0.29V28	–23.9V43	1.43	4.33	0.001
B[2]	0.46V27	–0.27V8	–0.28V28	0.053V29	–	5.99	4.53	0.001

[1] $r^2 = 0.510$; $r_1^2 = 0.375$; $r_2^2 = 0.052$; $r_3^2 = 0.040$; $r_4^2 = 0.027$; $r_5^2 = 0.014$
[2] $r^2 = 0.456$; $r_1^2 = 0.375$; $r_2^2 = 0.030$; $r_3^2 = 0.032$; $r_4^2 = 0.019$

tion rates and TP concentrations in the sediments, the model was formulated as:

$$[TP] = (L - S) / 0.956\, q_s$$

This version of the model was termed LCS-2.

Empirical calibration of TP retention (R_p) with data collected in oligotrophic Precambrian Shield lakes (Dillon and Kirchner 1975a) indicated that the settling velocity (v) averaged 12.4 m yr^{-1}. Employing this value, the LCS-1 model was tested on undeveloped Jerry Lake and Red Chalk Lake (main basin). Predicted TP concentrations agreed to within 0.1 mg m^{-3} with observed mean ice-free values. The TP concentrations predicted by LCS-2 did not agree as closely with the observed data for either Jerry Lake (10.6 mg m^{-3} vs 9.3 mg m^{-3} respectively) or the main basin of Red Chalk Lake.

Calibration of TP retention for the undeveloped east basin of Red Chalk Lake indicated that a settling velocity of 7.2 m yr^{-1} would be appropriate for lakes with anoxic hypolimnia. Therefore, the LCS-1 model was revised as follows:

$$[TP]_{IF} = L\,(1 - R_p) / 0.956\, q_s$$

where: $R_p = 12.4 / (12.4 + q_s)$ for oxic lakes
$R_p = 7.2 / (7.2 + q_s)$ for lakes with anoxic hypolimnia

D. PHOSPHORUS, CHLOROPHYLL *a*, PHOTOPLANKTON AND WATER CLARITY

1. TOTAL PHOSPHORUS – CHLOROPHYLL *a* RELATIONSHIP

Chlorophyll *a*, the principal photosynthetic pigment in most phytoplankton characteristic of oligo- and mesotrophic soft-water lakes, is widely accepted as a surrogate for algal biomass. Although the chlorophyll *a* content per unit of cell mass (or volume) may be highly variable (Nicholls and Dillon 1978), the ease of measurement combined with the fact that chlorophyll *a* represents a readily visible indication of trophic status, has resulted in the frequent use of chlorophyll *a* measurements rather than direct phytoplankton biomass measurements in limnological studies. For these reasons, chlorophyll *a* is used as a primary measure of trophic status.

A relationship for prediction of chlorophyll *a* concentration in lakes was first proposed by Dillon and Rigler (1974) who combined a global data set with information collected for 19 lakes in southern Ontario:

$$\log CHL\, a_{SS} = 1.45 \log TP_{SO} - 1.14 \quad (37)$$

This relationship was based on the observation made by Sakamoto (1966) that TP and chlorophyll[1] concentrations were highly correlated in a set of Japanese lakes with TN:TP>12 (by weight). Since Sakamoto noted that lakes with low TN:TP (<10) demonstrated better relationships between chlorophyll and TN than TP, Dillon and Rigler (1974) specified that the predictive relationship (equation 37) be used only in those cases where TN:TP>12.

In the subsequent decade, almost 60 predictive relationships relating chlorophyll *a* and nutrients have been reported (some are reviewed by Nicholls and Dillon 1978 and by Straškraba 1978). The many differences in these relationships are attributable to three facts:

(a) proponents of the various relationships have not been consistent with respect to analytical methods (chlorophyll *a*, chlorophyll *a* + *b* + *c*, and chlorophyll *a* corrected for phalopigments have all been used, often combined in a single relationship), sampling methodologies (volume-weighted vs. not volume-weighted samples, epilimnetic vs. euphotic zone vs. whole-lake samples), sampling times (annual vs. ice-free vs. stratification period), etc.,

(b) the chlorophyll content of algal cells may, as previously indicated, be quite variable. Chlorophyll may therefore only approximate algal biomass,

(c) the criterion that, for use of the TP-chlorophyll *a* relationship, the TN:TP ratio must be greater than 12 has often been ignored.

When model calibration has been restricted to consistent data sets and the original qualification concerning the TN:TP ratio has been observed, the results have been similar, supporting the premise that the TP-chlorophyll *a* relationship has considerable generality and predictive capability for north-temperate lakes.

Many of these relationships are not linear but predict that chlorophyll concentration increases faster than TP concentration. This has been explained by the fact that, as the TP input to lakes increases, the likelihood of anthropogenic contamination (almost invariably with readily biologically available phosphate-P) also increases. Therefore the proportion of the TP that is readily available for algal growth increases.

As a result of the many new relationships, significant improvements in predictability of chlorophyll *a* have been made, particularly by Smith and Shapiro (1981) and Smith (1982a, 1982b). These authors found (using global data sets with a wide variety of methodologies incorporated) that the chlorophyll *a* content (ice-free season) was dependent on the TN:TP ratio as well as the TP concentration of the lake itself:

$$\log CHL\, a_{IF} = 1.55\,(\log TP - \log(6.404/[0.0204\, TN/TP + 0.334])) \quad (38)$$

As the TN:TP ratio increases, its significance in predicting chlorophyll *a* decreases, such that if TN:TP>35, the role of nitrogen is insignificant. Subsequently, a revised relationship that produced a family of chlorophyll *a* – TP plots depending on the TN:TP ratio was proposed:

$$\log CHL\, a_{IF} = 0.653 \log TP + 0.548 \log TN - 1.517 \quad (39)$$

These models can be used for lakes with any TN:TP ratio, removing one of the qualifiers of the early relationships. Subsequently, Janus and Vollenweider (1981) and Canfield (1982) also demonstrated the importance of nitrogen; multivariate relationships including both TP and TN produced improved predictive capability in each case.

The TN:TP ratios of the study lakes are shown in Table 72. In all cases, they were >20 (usually >30), ranging up to ≃50 in both the ice-free and summer stratification periods. It was therefore expected that nitrogen would not play a major role in determining chlorophyll concentrations.

[1] Sakamoto's analytical method yields chlorophyll measurements that are greater than chlorophyll *a* but less than chlorophyll *a* + *b* + *c*.

Table 72. TN/TP (by weight[1]) averaged over the four-year (1976-79) period during the summer stratification (SS) and ice-free (IF) periods.

Lake	$(N/P)_{SS}$	$(N/P)_{IF}$
Blue Chalk	29.6	30.3
Chub	28.5	28.2
Dickie	24.0	26.7
Harp	38.5	39.9
Jerry	40.4	41.1
Red Chalk Main	51.8	52.3
Red Chalk East	42.6	44.1
Basshaunt	38.9	45.6
Bigwind	40.1	41.9
Buck	39.0	42.7
Crosson	30.5	31.1
Glen	21.6	24.7
Gullfeather	32.2	34.7
Little Clear	33.4	36.3
Solitaire	38.0	44.8
Walker	44.6	49.8

[1] Multiply by 2.21 for conversion to gram-atomic ratio

All of the aforementioned models were developed using data that covered a very wide range in parameter values; for example, Smith's most recent model (1982b) was developed from a data set (127 north-temperate lakes) with a range in chlorophyll *a* and TP concentrations of ≃ three orders of magnitude. The study lakes, and indeed all of the lakes in the Precambrian portion of southern Ontario with the exception of a few eutrophied lakes, span a much narrower range in trophic status. The range in chlorophyll *a* concentrations in either the ice-free or summer stratification periods in the "A" and "B" lakes considering individual years is only 1.4 – 6.0 $\mu g\ L^{-1}$ (Table 73). The four-year mean values (Table 74) vary by only a factor of 3 (1.8 – 5.5 $\mu g\ L^{-1}$), a range 300 times smaller than that in Smith's (1982a) or Dillon and Rigler's (1974) model. The variation in chlorophyll *a* content between years was not large, with the average coefficient of variation being 20% (range 6-36%, 6-44%) for both the ice-free and summer stratification periods. Mean concentrations for the four-year period were, therefore, used in subsequent model development.

Models for prediction of mean chlorophyll *a* concentration during both the ice-free and summer stratification periods were developed using the stepwise regression procedure. Independent variables that were available for selection in addition to TP included TN, several individual nitrogen forms, and the TN:TP ratio (because of the findings of Smith and others), pH (indicative of degree of acidification of the lakes), colour and DOC (measures of both the natural acidity of the lakes and the dissolved organic content of the lakes) and lake morpho metry (Table 75). For any lake the concentration of each

Table 74. Mean chlorophyll *a* concentration ($\mu g\ L^{-1}$) for the summer stratification (Chl a_{SS}) and ice-free periods (Chl a_{IF}). Coefficient of variation (c.v.) between years is expressed as %. Values are calculated as the mean of 4 annual means for the years 1976 to 1980.

Lake	Chl a_{SS}	(c.v.)	Chl a_{IF}	(c.v.)
Blue Chalk	1.83	(6)	1.82	(6)
Chub	3.01	(36)	3.01	(40)
Dickie	5.45	(9)	5.31	(12)
Harp	2.92	(9)	2.75	(10)
Jerry	3.16	(29)	3.09	(29)
Red Chalk Main	2.05	(9)	1.99	(8)
Red Chalk East	2.52	(13)	2.49	(12)
Basshaunt	1.78	(32)	1.74	(29)
Bigwind	2.64	(18)	2.47	(19)
Buck	2.03	(34)	2.27	(44)
Crosson	2.50	(31)	2.27	(28)
Glen	2.61	(30)	3.13	(10)
Gullfeather	4.82	(17)	4.56	(14)
Little Clear	2.32	(11)	2.15	(17)
Solitaire	1.80	(17)	1.72	(20)
Walker	2.39	(23)	2.27	(18)
		mean (20)		mean (20)

Table 73. Mean annual chlorophyll *a* concentration ($\mu g\ L^{-1}$) for the summer stratification (Chl a_{SS}) and ice-free periods (Chl a_{IF}) for the years 1976 to 1979.

	Chl a_{SS}				Chl a_{IF}			
Lake	1976	1977	1978	1979	1976	1977	1978	1979
Blue Chalk	1.88	1.89	1.68	1.87	1.87	1.85	1.67	1.89
Chub	3.33	1.98	4.41	2.34	3.14	1.99	4.65	2.26
Dickie	5.97	5.03	5.10	5.73	4.51	5.85	5.79	5.10
Harp	3.01	2.92	2.55	3.21	2.71	2.78	2.42	3.08
Jerry	2.03	2.85	3.70	4.06	2.05	2.73	3.48	4.10
Red Chalk Main	2.26	1.81	2.12	2.01	2.10	1.75	2.09	2.01
Red Chalk East	–[1]	2.26	2.89	2.42	–[1]	2.24	2.83	2.39
Basshaunt	2.00	1.00	1.80	2.32	1.84	1.07	1.78	2.29
Bigwind	2.07	2.50	3.18	2.82	1.84	2.64	2.94	2.46
Buck	2.05	1.32	1.79	2.98	1.94	1.43	1.97	3.74
Crosson	3.50	1.80	2.72	1.98	3.04	1.66	2.55	1.85
Glen	1.78	–[1]	2.72	3.33	3.05	3.00	2.90	3.57
Gullfeather	4.94	4.48	5.92	3.96	4.67	4.99	4.93	3.66
Little Clear	2.41	2.50	2.43	1.94	2.56	2.06	2.31	1.69
Solitaire	2.18	1.82	1.42	1.78	2.19	1.77	1.40	1.51
Walker	2.33	1.78	2.35	3.12	2.28	1.80	2.20	2.81

[1] no data

of the chemical variables can be predicted (TP) using the methodologies outlined in this report or easily measured (other than TP) during the appropriate time period. Any model must, of course, be applied for predictive purposes with extreme caution to cases where the independent parameters are less than or greater than the range used in model construction. These ranges are also shown in Table 75.

The most significant correlations between mean chlorophyll *a* concentration in the study lakes ("A" and "B" lakes) and independent chemical variables are shown in Table 76. For both the ice-free and summer stratification periods, TP concentration (epilimnetic) was the strongest correlate; however, strong correlations also existed with other TP parameters (volume-weighted epilimnetic plus metalimnetic concentration), with several nitrogen parameters and with DOC and colour. Although the range in chlorophyll *a* concentration among lakes was extremely small (3-fold), TP (epilimnetic) concentration explained 77% of the variance in each time period.

Stepwise regression models for the two time periods are shown in Table 77. The second step in each model (highly significant model improvement each time) was the entry of the TN:TP ratio (epilimnetic). Although this is consistent with the findings of Smith and Shapiro (1981) and Smith (1982a), it is surprising because the ranges in both chlorophyll *a* and TN:TP were small (three-fold and two-fold, respectively). Other nitrogen parameters were selected in subsequent steps, again emphasizing that nitrogen and phosphorus, not phosphorus alone, determine chlorophyll *a* standing stock. The complete models explained 92and 88% of the variance in chlorophyll *a* in the ice-free and summer stratification periods respectively.

For predictive purposes, truncated (2-step) models are presented in Table 78. Using only TP (epilimnetic) and the TN:TP ratio (epilimnetic), 82 and 84% of the variance in chlorophyll *a* in the two time periods is explained. The resultant models are:

$$\mathrm{Chl}\ a_{SS} = 0.51\ \mathrm{TP}_{EP} + 0.059\ (\mathrm{N/P})_{EP} - 3.35 \qquad (40)$$

$$\mathrm{Chl}\ a_{IF} = 0.48\ \mathrm{TP}_{EP} + 0.060\ (\mathrm{N/P})_{EP} - 3.14 \qquad (41)$$

To facilitate use of the models by eliminating the need for measurement of TN:TP in the epilimnion over the entire summer stratification period for a particular lake, the relationship

Table 75. Independent variables used in the generation of chlorophyll *a* models. For each parameter, epilimnetic (EP), metalimnetic (ME), summer stratification (SS), ice-free period (IF), ice-covered period (IC), spring overturn (SO), fall overturn (FO), and annual average (AN) data were used. For TP volume-weighted epilimnetic plus metalimnetic (EM) data were also used. Ranges in the epilimnion, and the whole lake during the ice-free and annual periods are shown. Units are mg m^{-3} unless specified.

		Range		
Parameter	Symbol	EP	IF	AN
total phosphorus	TP	4.9-13.0	5.6-19.3	5.7-20.7
nitrate (+nitrite)[1]	NO_3	8.7-111	16-133	18-138
ammonium	NH_4	9.0-20	12-147	14-158
total Kjeldahl nitrogen	TKN	190-310	210-440	210-460
total organic nitrogen ($TKN - NH_4$)	TON	180-290	190-300	190-310
total inorganic nitrogen (NO_3+NH_4)	TIN	19-127	35-190	40-220
total nitrogen ($TKN+NO_3$)	TN	200-410	220-480	230-520
total nitrogen/ total phosphorus	N/P	25.8-54.7	24.7-52.3	25.3-53.7
inorganic nitrogen/ total phosphorus	I/P	2.6-9.8	4.7-17.9	5.7-18.4
pH (–)	PH	5.86-7.83	5.58-7.39	5.58-7.23
colour (Hazen units)	COL	<5.0-12.4	5.0-22.9	5.2-24.4
dissolved organic carbon (mg L^{-1})	DOC	2.3-5.8	2.7-6.7	2.8-6.9
lake area (10^4 m^2)	A		10.9-124	
lake volume (10^6 m^3)	V		0.75-16.4	
mean depth (m)	Z		4.8-16.7	
maximum depth (m)	ZM		12.0-40.0	

[1] nitrite was always undetectable when measured separately

Table 76. Correlations between mean chlorophyll *a* concentration (euphotic zone) during summer stratification (SS) and the ice-free period (IF) and independent chemical variables (listed in Table 75).

Chl a_{SS}			Chl a_{IF}		
variable	r[1]	F[2]	variable	r[1]	F[2]
TP_{EP}	0.878	47.0	TP_{EP}	0.870	43.4
DOC_{AN}	0.854	37.7	TP_{EM}	0.868	42.8
TP_{EM}	0.852	37.0	TON_{EP}	0.832	31.4
TN_{EP}	0.806	25.9	TKN_{EP}	0.826	30.0
TON_{EP}	0.797	24.3	DOC_{AN}	0.817	28.2
TKN_{EP}	0.791	23.4	TN_{EP}	0.806	25.9
COL_{IC}	0.770	20.4			

[1] correlation coefficient
[2] partial F-value

Table 77. Stepwise regression models for the prediction of Chl *a* concentration in the euphotic zone during the summer stratification (SS) and ice-free (IF) periods. Independent variables and their units are listed in Table 75.

	Step 1	Step 2	Step 3	Step 4	Constant	p	SE
$(\mathrm{Chl}\ a)_{SS}$[1]	0.59 TP_{EP}	0.082 $(N/P)_{EP}$	−0.0063 TIN_{IF}	–	−4.18	<0.001	0.394
$(\mathrm{Chl}\ a)_{IF}$[2]	0.83 TP_{EP}	0.12 $(N/P)_{EP}$	−0.0018 TIN_{EP}	−0.0076 TON_{IC}	−5.56	<0.001	0.330

[1] $r^2 = 0.882$; $r_1^2 = 0.771$; $r_2^2 = 0.064$; $r_3^2 = 0.047$
[2] $r^2 = 0.919$; $r_1^2 = 0.756$; $r_2^2 = 0.064$; $r_3^2 = 0.061$; $r_4^2 = 0.038$

$$(TN/TP)_{EP} = 0.43\,(TN/TP)_{FO} + 19.4 \quad (42)$$

may be used. This relationship ($r^2 = 0.58$; $p < 0.05$) requires measurement of the TN:TP ratio only at fall overturn, i.e., one or two measurements. A similar relationship based on the spring overturn period was less useful.

Table 78. Stepwise regression models truncated after the second step for the prediction of mean Chl *a* concentration in the euphotic zone during summer stratification (SS) and ice-free (IF) periods. Independent variables and their units are listed in Table 75.

	Step 1	Step 2	Constant	p	SE
$(Chl\ a)_{SS}$[1]	$0.51\ TP_{EP}$	$0.059\ (N/P)_{EP}$	–3.35	<0.001	0.451
$(Chl\ a)_{IF}$[1]	$0.48\ TP_{EP}$	$0.060\ (N/P)_{EP}$	–3.14	<0.001	0.453

[1] $r^2 = 0.835$; $r_1^2 = 0.771$; $r_2^2 = 0.064$
[2] $r^2 = 0.820$; $r_1^2 = 0.756$; $r_2^2 = 0.064$

2. PHYTOPLANKTON COMPOSITION, BIOMASS AND TOTAL P

It is generally assumed that increases in nutrient levels in lake waters will increase total phytoplankton biomass (PB) and change the taxonomic composition of the phytoplankton community.

A total of 177 phytoplankton genera were observed in the 16 lakes in the four years (Nakamoto et al. 1983). The highest proportional contribution was from the green algae, Chlorophyceae (32 genera). The total number of genera in each lake (four-year averages) ranged from 38 for Glen Lake to 82 for Red Chalk Lake East and was not correlated with TP (after correcting for numbers of samples examined).

Based on the genus data (177 genera × 16 lakes × 4 years) a clustering analysis of dissimilarity coefficients among the lakes showed much greater year-to-year than lake-to-lake differences. Lakes in 1976 were alike, but different from the lakes in 1977-78, and different again from the lakes in 1979. Some of this variability might have been related to the rate of warming of the lakewater in early summer. The local air temperatures during April-May (Muskoka airport) were most similar during 1977-78 (98% similarity) compared with 1976-79 (95% similarity).

Based on both the total phytoplankton biomass (PB) and the proportional contribution to PB of the seven dominant algal classes, there were marked differences in calculated percentage similarities among the lakes. Dickie and Gullfeather lakes were similar to each other but unlike any of the other lakes. Jerry Lake also was unlike most of the other lakes and was similar to only Red Chalk Main, Glen and Harp Lakes at a level greater than 80%. In contrast, Crosson Lake had the highest average level of similarity with all other lakes and showed similarity levels exceeding 80% with all lakes except Jerry, Gullfeather and Dickie (Figure 19).

Noxious blooms of blue-green algae have been reported in hard-water (Nicholls et al. 1983) and in nutrient-enriched soft-water lakes in Ontario (Dillon et al. 1978, Schindler and Fee 1974). Such blooms may render water toxic to animals, farm animals for example, that may normally consume the water, and also drastically impair aesthetic and recreational appeal of lakes.

The proportion of PB of the lakes contributed by blue-greens was very low (4-year ice-free period averages ranged from 0.1-4.1%), and was not significantly correlated with either TP or the ratio of TN:TP. Data from other Ontario lakes suggest that the significant occurrence (>0.2 mm^3/L) of blue-green bloom-forming algae is not likely in lakewaters with TN:TP mass ratios greater than 25. The TN:TP values for the study lakes (4-year ice-free period means) ranged between 25 and 52 (Table 72).

Composition of phytoplankton communities may be influenced by light. Hence, differences in community structure among the LCS lakes might be related to differences in relative illuminance of their epilimnia. The effective light climate ($\bar{I}$ in Table 79) was apparently an

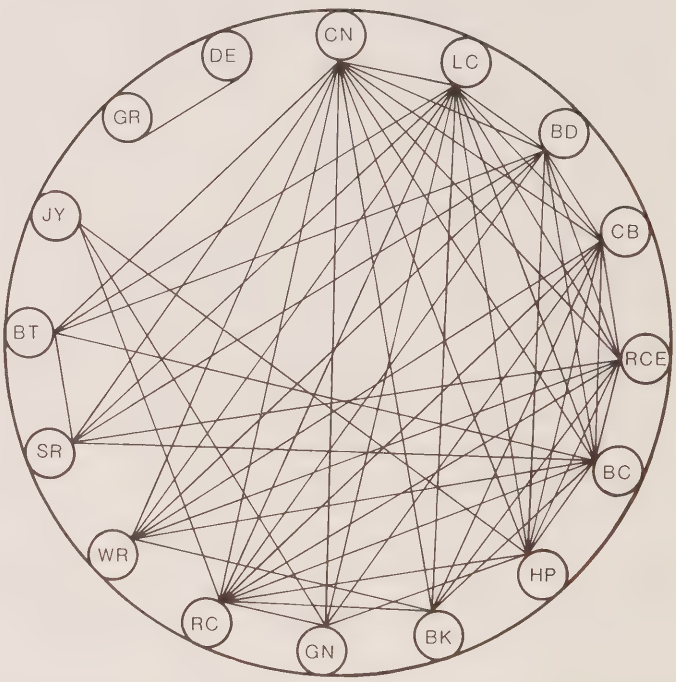


Figure 19 Similarity of phytoplankton communities among the 16 Muskoka-Haliburton lakes based on a 1:1 weighting of average phytoplankton biomass (1976-79) and the proportionality of algal class composition. Lakes joined by a straight line indicate a similarity >80%. The lakes are arranged in clockwise order of highest (Crosson – CN) to lowest (Dickie – DE) average percentage similarity with all other lakes in the set.

BC — Blue Chalk
BD — Bigwind
BK — Buck
BT — Basshaunt
CB — Chub
CN — Crosson
DE — Dickie
GN — Glen
GR — Gullfeather
HP — Harp
JY — Jerry
LC — Little Clear
RC — Red Chalk
RCE — Red Chalk East
SR — Solitaire
WR — Walker

important factor in determining the composition of the lakes' phytoplankton. The importance (mg algae per mg TP) of diatoms and green algae in the clear water (high $\bar{I}$) lakes and of the Chrysophyceae and Dinophyceae in the coloured lakes (low $\bar{I}$) confirms the experimental evidence of Wall and Briand (1979) which showed that diatoms and green algae preferred higher light intensities than certain flagellates.

Table 79. Average biomass of phytoplankton per unit total phosphorus (BP/TP) and average effective light climate ($\bar{I}$)[1] for the epilimnia of the "A" lakes during 1979.

	PB/TP (mm³/g)		$\bar{I}$ (cal/cm².day)	
Lake	June-July	Aug.-Sept.	June-July	Aug.-Sept.
Blue Chalk	0.12	0.072	130	65
Red Chalk – Main	0.098	0.126	140	70
Red Chalk – East	0.22	0.076	129	72
Chub	0.052	0.052	104	52
Dickie	0.128	0.151	87	44
Harp	0.118	0.119	112	61
Jerry	0.07	0.106	123	61
Mean ± St. D.	0.10± 0.03[2]	0.11[5]± 0.005[2]	118±18[3]	61±10[3]

[1] $\bar{I} = \dfrac{I_o^1\,(1-e^{kz})}{kz}$

where – I_o^1 is downwelling photosynthetically available radiation at the lake surface estimated as 41% of total downwelling radiation (200-3000nm) measuring in the field.

– k is the apparent vertical extinction coefficient of downwelling photosynthetically available radiation. This was estimated as 1.45/Secchi transparency depth (Walker 1978)

– z is measured epilimnetic depth

[2] not significantly different (t-test; $p > 0.05$)

[3] significantly different (t-test; $p < 0.05$)

The equation relating Cryptophyceae plus Dinophyceae importance (units as above) to $\bar{I}$ (cal/cm².day) was:

$$\text{Crypto} + \text{Dino} = -0.35\,\bar{I} + 43.7 \; (r = 0.76; P < 0.01) \quad (43)$$

and the equation relating *Cyclotella* (a diatom) and Chlorophyceae to $\bar{I}$ (cal/cm².day) was:

$$\textit{Cyclotella} + \text{Chloro} = 0.42^{0.04\bar{I}} \; (r = 0.62; P < 0.05) \quad (44)$$

The mean phytoplankton biomass measured in the ice-free and summer stratification periods in the "A" and "B" lakes is shown in Table 80. Like chlorophyll *a*, the range in phytoplankton biomass among lakes was very small. The four-year mean concentrations (Table 80) ranged from only 0.32 mg L^{-1} (Basshaunt Lake) to 1.08 mg L^{-1} (Dickie Lake) during summer stratification. However, with the average coefficient of variation between years being 30% (Table 81), the differences were greater than for chlorophyll *a*.

Analysis of variance and least significant difference tests showed that the average ice-free season estimate of phytoplankton biomass of all lakes for any one of the four years was not significantly different from that of any other year. The values were 0.57, 0.54, 0.65 and 0.58 mg³/L for 1976, 1977, 1978 and 1979, respectively.

Unlike chlorophyll *a*, phytoplankton biomass has been infrequently used as a measure of the trophic status of lakes. This may be largely attributed to the difficulty in measurement of biomass compared to chlorophyll *a*.

Few predictive models for phytoplankton biomass have been proposed. Nicholls and Dillon (1978) demonstrated that variance in a TP-chlorophyll *a* model for relatively hard-water lakes in southern Ontario could be substantially reduced if chlorophyll *a* was replaced by phytoplankton biomass. Kalff and Knoechel (1978) developed a predictive relationship with TP concentration

Table 80. Mean phytoplankton biomass (mg L^{-1}) for the summer stratification (PB_{SS}) and ice-free (PB_{IF}) periods for the years 1976 to 1979.

	PB_{SS}				PB_{IF}			
Lake	1976	1977	1978	1979	1976	1977	1978	1979
Blue Chalk	0.34	0.37	0.42	0.37	0.33	0.36	0.39	0.40
Chub	0.79	0.36	0.59	0.43	0.83	0.38	0.77	0.42
Dickie	0.86	0.72	0.92	1.81	0.66	0.90	1.11	1.62
Harp	0.75	0.66	0.69	0.66	0.72	0.64	0.62	0.64
Jerry	0.35	0.76	1.41	0.66	0.38	0.75	1.36	0.66
Red Chalk Main	0.96	0.56	0.51	0.42	0.92	0.54	0.48	0.41
Red Chalk East	–[1]	0.53	0.49	0.52	–[1]	0.50	0.47	0.50
Basshaunt	0.21	–[1]	0.40	0.36	0.21	–[1]	0.37	0.43
Bigwind	0.34	0.78	0.54	0.32	0.31	0.72	0.52	0.28
Buck	0.73	0.18	0.43	0.43	0.73	0.24	0.44	0.59
Crosson	0.91	0.25	0.78	0.27	0.67	0.22	0.69	0.25
Glen	0.50	–[1]	1.07	0.87	0.50	–[1]	1.07	0.72
Gullfeather	0.87	0.85	1.04	0.92	0.93	1.18	1.01	0.83
Little Clear	0.53	0.49	0.23	0.62	0.54	0.51	0.23	0.52
Solitaire	0.30	0.37	0.33	0.37	0.30	0.38	0.30	0.34
Walker	0.42	0.27	0.66	0.66	0.35	0.31	0.57	0.58

[1] no data

using a small global data set. Schindler et al. (1978) found very strong correlations between algal biomass (or volume) and TP (epilimnetic or whole-lake) concentration, as well as with several functions combining TP load, lake morphometry and lake water balance, for a number of lakes in the Experimental Lakes Area, Ontario.

Correlations between phytoplankton biomass and chemical variables are shown in Table 82. Total organic nitrogen (epilimnetic) and TP (epilimnetic plus metalimnetic) were the best correlates of phytoplankton biomass during the summer stratification and ice-free periods respectively; however, in each case, other phosphorus and nitrogen parameters were almost equally highly correlated. Using stepwise regression analyses with F ratios to add or delete independent variables set at 90%, one-variable equations were formulated (Table 83); there were no significant additions at the second step. These models explained 62 and 63% of the variance in phytoplankton biomass in the summer stratification and ice-free periods, substantially less than was explained for chlorophyll *a*. This result was unexpected since chlorophyll *a* is considered to be only an approximation of algal biomass, the parameter that is anticipated to be highly correlated with nutrient levels.

Of the components of the algal biomass, the diatoms (Bacillariophyceae) comprise a significant portion in many soft-water oligotrophic and mesotrophic lakes.

Table 81. Mean phytoplankton biomass (mg L^{-1}) for the summer stratification (PB_{SS}) and ice-free (PB_{IF}) periods. Coefficient of variation (c.v.) between years is expressed as %. Values are calculated as the mean of 4 annual means for the years 1976 to 1980.

Lake	PB_{SS}	(c.v.)	PB_{IF}	(c.v.)
Blue Chalk	0.38	(9)	0.37	(9)
Chub	0.54	(35)	0.60	(39)
Dickie	1.08	(46)	1.07	(38)
Harp	0.69	(6)	0.66	(7)
Jerry	0.80	(56)	0.79	(52)
Red Chalk Main	0.61	(39)	0.59	(39)
Red Chalk East	0.51	(4)	0.49	(4)
Basshaunt	0.32	(31)	0.34	(33)
Bigwind	0.50	(43)	0.46	(45)
Buck	0.44	(51)	0.50	(42)
Crosson	0.55	(62)	0.46	(56)
Glen	0.81	(36)	0.76	(38)
Gullfeather	0.92	(9)	0.99	(15)
Little Clear	0.47	(36)	0.45	(33)
Solitaire	0.34	(10)	0.33	(12)
Walker	0.50	(38)	0.45	(32)
		mean (32)		mean (31)

Table 82. Correlations between mean total phytoplankton biomass in the euphotic zone during summer stratification (SS) and the ice-free (IF) period and independent chemical variables (listed in Table 75).

PB_{SS}			PB_{IF}		
variable	r[1]	F[2]	variable	r[1]	F[2]
TON_{EP}	0.789	22.6	TP_{EM}	0.800	24.1
TP_{EM}	0.777	21.3	TON_{EP}	0.784	22.3
TKN_{EP}	0.773	20.7	TKN_{EP}	0.772	20.6
TN_{EP}	0.748	17.8	TN_{EP}	0.771	20.5
TP_{EP}	0.729	15.9	TP_{EP}	0.767	20.0

[1] correlation coefficient
[2] partial F-value

Table 83. Stepwise regression models for the prediction of mean total phytoplankton biomass (PB) in the euphotic zone during the summer stratification (SS) and ice-free (IF) periods. Independent variables and their units are listed in Table 75.

	Step 1	Constant	p	SE
PB_{SS}[1]	0.0052 TON_{EP}	−0.64	<0.001	0.137
PB_{IF}[2]	0.068 TP_{EM}	−0.0044	<0.001	0.139

[1] $r^2 = 0.617$ [2] $r^2 = 0.633$

Table 84. Mean corrected phytoplankton biomass (less diatom biomass) (mg L^{-1}) for the summer stratification (CPB_{SS}) and ice-free (CPB_{IF}) periods for the years 1976 to 1979.

	CPB_{SS}				CPB_{IF}			
Lake	1976	1977	1978	1979	1976	1977	1978	1979
Blue Chalk	0.17	0.25	0.25	0.13	0.16	0.24	0.22	0.18
Chub	0.35	0.29	0.44	0.39	0.41	0.30	0.70	0.38
Dickie	0.83	0.69	0.73	1.42	0.56	0.82	0.87	1.28
Harp	0.33	0.29	0.43	0.36	0.33	0.29	0.38	0.35
Jerry	0.19	0.28	0.35	0.55	0.23	0.28	0.32	0.54
Red Chalk Main	0.20	0.15	0.29	0.20	0.22	0.16	0.29	0.40
Red Chalk East	–[1]	0.26	0.44	0.40	–[1]	0.25	0.42	0.39
Basshaunt	0.20	–[1]	0.18	0.21	0.20	–[1]	0.18	0.33
Bigwind	0.18	0.18	0.40	0.21	0.16	0.25	0.38	0.20
Buck	0.18	0.17	0.23	0.20	0.20	0.21	0.28	0.41
Crosson	0.36	0.20	0.33	0.20	0.30	0.19	0.29	0.20
Glen	0.34	–[1]	0.63	0.43	0.34	–[1]	0.52	0.40
Gullfeather	0.69	0.75	0.89	0.85	0.68	0.87	0.85	0.75
Little Clear	0.22	0.26	0.18	0.28	0.27	0.32	0.20	0.27
Solitaire	0.15	0.25	0.23	0.22	0.15	0.27	0.22	0.22
Walker	0.26	0.23	0.38	0.62	0.23	0.27	0.40	0.55

[1] no data

Since a large proportion of many diatom species consists of a vacuole (empty space) while the cytoplasmic portion of the cell is highly variable, and since it is often difficult to distinguish between living and dead diatoms, the total phytoplankton biomass excluding diatoms (called the "corrected phytoplankton biomass") was calculated (Table 84). In many cases, these values were substantially lower than the total phytoplankton biomass (e.g., Red Chalk Lake, Main Basin – 0.21 vs. 0.61 mg L^{-1} in the summer stratification period). The variability between years (Table 85) was comparable (coefficient of variation 27 and 28% in the two time periods) to that of total phytoplankton biomass.

The correlations between the corrected phytoplankton biomass and nutrient parameters (Table 86) were much better than when total phytoplankton biomass was used (Table 82). In both time periods the epilimnetic and combined epilimnetic and metalimnetic TP concentrations were the best correlates, with several nitrogen fractions and DOC also strongly correlated. In neither instance was pH significantly correlated.

Stepwise regression models were constructed for corrected phytoplankton biomass using the epilimnetic TP concentration as the first step in each time period for consistency (Table 87). As was the case for chlorophyll *a*, the TN:TP ratio was the second parameter added to each model. The addition of this parameter resulted in very significantly increased predictive capability of the models. With all variables explaining significant amounts of variance of CPB (five in the summer stratification period, four in the ice-free period), the regression models explained 98 and 93% of the variance in corrected phytoplankton biomass, despite the narrow range in biomass. Truncated models (Table 88) which require less complete data accounted for 82 and 84% of the variance in corrected phytoplankton biomass (CPB).

The resultant models are:

$$CPB_{SS} = 0.108TP_{EP} + 0.015(TN/TP)_{EP} - 1.05 \qquad (45)$$

$$CPB_{IF} = 0.098TP_{EP} + 0.012(TN/TP)_{EP} - 0.84 \qquad (46)$$

In summary, phytoplankton biomass excluding diatoms can be predicted much more successfully than total phytoplankton biomass, and slightly more successfully than chlorophyll *a*. The phytoplankton concentration (either biomass or chlorophyll *a*) in these lakes is controlled by nutrient, particularly TP, concentrations, not by the pH of the lakewater.

Table 85. Mean corrected phytoplankton biomass (less diatom biomass), for the summer stratification (CPB_{SS}) and ice-free (CPB_{IF}) periods. Coefficient of variation (c.v.) between years is expressed as %. Values are calculated as the mean of 4 annual means for the years 1976 to 1980.

Lake	CPB_{SS}	(c.v.)	CPB_{IF}	(c.v.)
Blue Chalk	0.20	(30)	0.20	(18)
Chub	0.37	(17)	0.45	(39)
Dickie	0.92	(37)	0.88	(34)
Harp	0.35	(17)	0.34	(11)
Jerry	0.34	(45)	0.34	(40)
Red Chalk Main	0.21	(28)	0.27	(38)
Red Chalk East	0.37	(26)	0.35	(26)
Basshaunt	0.20	(7)	0.23	(35)
Bigwind	0.24	(44)	0.25	(38)
Buck	0.20	(13)	0.28	(35)
Crosson	0.27	(31)	0.25	(23)
Glen	0.47	(32)	0.42	(22)
Gullfeather	0.80	(11)	0.79	(11)
Little Clear	0.24	(18)	0.27	(18)
Solitaire	0.21	(21)	0.22	(22)
Walker	0.37	(48)	0.36	(40)
	mean	(27)	mean	(28)

Table 86. Correlations between mean phytoplankton biomass excluding diatoms (CPB) in the euphotic zone during summer stratification (SS) and the ice-free (IF) periods. Independent variables and their units are listed in Table 75.

CPB_{SS} variable	r^1	F^2	CPB_{IF} variable	r^1	F^2
TP_{EM}	0.862	40.6	TP_{EP}	0.875	45.9
TP_{EP}	0.847	35.6	TP_{EM}	0.869	43.3
TON_{EP}	0.806	26.0	DOC_{AN}	0.805	25.7
TKN_{EP}	0.804	25.6	TN_{EP}	0.799	24.7
TN_{EP}	0.803	25.4	TKN_{EP}	0.785	22.4
DOC_{AN}	0.780	21.8	TON_{EP}	0.785	22.4
			COL_{IC}	0.773	20.7

[1] correlation coefficient
[2] partial F-value

Table 88. Stepwise regression models truncated after the second step for the prediction of mean phytoplankton biomass excluding diatoms (CPB) in the euphotic zone during summer stratification (SS) and the ice-free (IF) periods. Independent variables and their units are listed in Table 75.

	Step 1	Step 2	Constant	p	SE
CPB_{SS}[1]	0.108 TP_{EP}	0.015 $(N/P)_{EP}$	–1.05	<0.001	0.097
CPB_{IF}[2]	0.098 TP_{EP}	0.012 $(N/P)_{EP}$	–0.84	<0.001	0.084

[1] $r^2 = 0.819$; $r_1^2 = 0.718$; $r_2^2 = 0.101$
[2] $r^2 = 0.841$; $r_1^2 = 0.766$; $r_2^2 = 0.075$

Table 87. Stepwise regression models for the prediction of mean phytoplankton biomass excluding diatoms (CPB) in the euphotic zone during summer stratification (SS) and the ice-free (IF) periods. Independent variables and their units are listed in Table 75.

	Step 1	Step 2	Step 3	Step 4	Step 5	Constant	p	SE
CPB_{SS}[1,3]	0.31 TP_{EP}	0.070 $(N/P)_{EP}$	–0.10 $(I/P)_{EP}$	–0.0066 TKN_{EP}	0.76 pH_{IF}	–2.97	<0.001	0.039
CPB_{IF}[2]	0.25 TP_{EP}	0.039 $(N/P)_{EP}$	–0.0053 TN_{EP}	0.080 pH_{EP}	–	–2.10	<0.001	0.059

[1] $r^2 = 0.978$; $r_1^2 = 0.718$; $r_2^2 = 0.101$; $r_3^2 = 0.077$; $r_4^2 = 0.037$; $r_5^2 = 0.027$
[2] $r^2 = 0.933$; $r_1^2 = 0.766$; $r_2^2 = 0.075$; $r_3^2 = 0.070$; $r_4^2 = 0.022$
[3] two additional significant steps ($NO2_{AN}$, TON_{IC}) contributed slightly to r^2

3. EFFECTS OF ZOOPLANKTON BIOMASS ON PHOSPHORUS-PHYTOPLANKTON RELATIONSHIPS

Standing stocks of phytoplankton are determined by rates of algal growth, a phosphorus-dependent process, and by rates of loss or removal of algae from the water column. Zooplankton grazing may be the most important removal mechanism. Explicit consideration of differences among lakes in expressions of zooplankton community grazing pressure may, therefore, result in improvements in predictability of phytoplankton biomass.

Direct measurements of herbivorous zooplankton grazing or filtering rates were not made in the study lakes. As filtering rates (to be discussed in detail) of individual zooplankters are strongly correlated with animal size (Burns 1969), community filtering rates should be positively correlated with community biomass, the product of zooplankton community density (animals L^{-1}) and average zooplankton size. In this section whether predictions of water clarity or phytoplankton biomass in the study lakes may be improved by inclusion of total crustacean (B_z) or herbivorous crustacean (B_h) zooplankton biomass in regression models is examined. Classification of animals as herbivores or carnivores followed Sprules (1980).

Zooplankton were enumerated in volume-weighted whole lake composite samples. Average biomass values generated from these samples are given in Table 89. Averages of B_z for the ice-free season ranged from 27.3 to 88.4 mg m^{-3} (dry weight) among the "A" lakes from 1977 to 1979. The average for all lakes was 48.7 mg m^{-3}. Among the "B" lakes, B_z ranged from 20.3 to 125.1 mg m^{-3} with a mean of 64.1 mg m^{-3}. B_h averaged 74% of B_z.

B_z and B_h were added to the set of independent variables used for model development. The stepwise regression procedure was then repeated using ice-free season and summer stratification season averages of Secchi depth, chlorophyll *a*, phytoplankton biomass, and corrected phytoplankton biomass as dependent variables. Neither B_z nor B_h explained a sufficient amount of the residual variance in any of the dependent variables to be included in any of the regression models.

The regression analyses were then repeated for all dependent variables excluding Secchi transparency, with first B_z then B_h inserted as the second independent variable. Decision criteria for choice of the first independent variable were unchanged. In a second exercise, B_z and B_h were also inserted as the third independent variable. In all cases, insertion of B_z or B_h resulted in no significant ($p > 0.10$) reduction in residual variance in chlorophyll *a* or phytoplankton biomass (Table 90).

These analyses should not be interpreted as signifying that zooplankton are never important in any particular lake or at any particular time in controlling phytoplankton biomass. Recent changes in *Daphnia* populations in Lake Washington in the U.S.A., for example, have had substantial effects on the phytoplankton of the lake (Edmundson and Litt 1982). Nevertheless, the analyses do indicate that explicit consideration of average

Table 90. Examples of changes in coefficient of determination (r^2) for regressions produced by selection of either B_z or B_h as the second independent variable in multiple linear regression models. Note that the increase in r^2 is small (and not statistically significant) in each case.

Dependent Variable[1]	First Independent Variable		r^2 with	
	Name	r^2	B_z	B_h
Chlor a_{IF}	TP_{EP}	0.756	0.765	0.759
Chlor a_{SS}	TP_{EP}	0.771	0.795	0.787
PB_{IF}	TP_{EM}	0.633	0.693	0.682
PB_{SS}	TON_{EP}	0.617	0.618	0.617
CPB_{IF}	TP_{EP}	0.766	0.773	0.769
CPB_{SS}	TP_{EP}	0.718	0.719	0.721

[1] PB is total phytoplankton biomass; CPB is total phytoplankton biomass less diatom biomass.

Table 89. Ice-free season averages of total (B_z) and herbivorous (B_h) crustacean zooplankton biomass in the study lakes for 1977, 1978 and 1979. Arithmetic means of 1977-1979 are also included.

Lake	1977		1978		1979		mean	
	B_z	B_h	B_z	B_h	B_z	B_h	B_z	B_h
Blue Chalk	44.7	36.3	56.0	39.4	69.9	51.5	56.9	42.4
Chub	47.6	30.4	49.1	26.6	44.8	36.6	47.2	31.2
Dickie	88.4	80.7	28.3	21.4	43.9	27.4	53.5	43.1
Harp	33.9	27.1	39.5	27.1	47.1	36.1	40.2	30.1
Jerry	27.3	18.1	43.2	26.4	73.7	44.3	48.1	29.6
Red Chalk Main	30.6	21.4	37.1	23.7	43.7	25.1	37.1	23.4
Red Chalk East	37.3	27.7	75.6	56.6	61.8	45.7	58.2	43.3
Basshaunt	–	–	46.9	27.0	60.0	46.1	53.5	36.5
Bigwind	41.1	35.5	50.4	42.2	55.0	45.6	48.8	41.1
Buck	20.7	14.7	34.0	26.1	42.7	36.7	32.5	25.8
Crosson	70.3	36.4	74.2	62.3	76.2	59.4	73.5	52.7
Glen	–	–	108.7	86.8	99.7	88.2	104.2	87.5
Gullfeather	36.6	29.9	125.1	93.3	123.5	99.1	95.1	74.1
Little Clear	65.6	49.5	83.8	56.8	63.4	55.8	70.9	54.0
Solitaire	36.0	23.5	47.2	32.4	67.7	51.8	50.3	36.0
Walker	20.3	17.0	60.4	51.0	64.8	50.6	48.5	39.5

total or herbivorous zooplankton biomass does not significantly improve the predictability of long-term average phytoplankton biomass for relatively nutrient poor lakes in the Muskoka-Haliburton area.

It is assumed for this analysis that zooplankton biomass was a good correlate of zooplankton community filtering rate. This assumption implies that filtering rates per unit biomass should be constant among species. As Egloff and Palmer (1974) indicated, this may not be true, the analyses were repeated using estimated community filtering rates.

Zooplankton filtering rates (FR) can be conceptualized as that amount of lake water (for individual zooplankters) or that proportion of the lake (for the zooplankton community) which is effectively swept clear of prey by the grazing activities of the animals in unit time. Typically, individual FR is expressed in units of mL animal^{-1} d^{-1} and community FR in units of mL of cleared lake water per L of lake water per day or in % (of the lake) d^{-1}.

FR was estimated for individual zooplankters as follows. Average lengths of all common herbivores were calculated from their measured average weights using literature length-weight relationships. FR was calculated from length-FR relationships developed by Chow (unpub. studies) for species commonly found in Ontario lakes. FR ranged from 0.28 mL animal^{-1} d^{-1} for *Eubosmina tubicen* to 24.09 mL animal^{-1} d^{-1} for *Daphnia rosea*. The mean FR was 5.3 mL animal^{-1} d^{-1} (Table 91). This calculated value is similar to measured values in Ontario lakes (Haney 1973).

It was previously assumed that B_z and B_h were excellent correlates of community FR, i.e., that FR/W, where W is the average weight (μg animal^{-1}) of an individual of a taxon, would be identical for all taxa. The calculated values of FR/W for each taxon ranged over an order of magnitude, however, from 0.7 mL μg^{-1} d^{-1} or less for *B. longirostris, Chydorus* spp., *E. longispina* and *E. tubicen* to about 5 mL μg^{-1} d^{-1} for nauplii (Table 91). This suggests that the assumption of constant FR/W might (recall FR was estimated not measured) have been in error and that B_z and B_h may not be good estimators of community FR, as species composition of zooplankton does vary among the study lakes.

Estimates of community FR were, therefore, generated for the ice-free season for each lake. FR was taken to be the sum for all taxa of the product of FR for individuals of each taxon (Table 92) and the densities of those animals (animals L^{-1}). Calculated community FR ranged from 1.96% d^{-1} in 1977 in Buck Lake to 23.2% d^{-1} in 1979 in Glen Lake. The long-term means (1977 to 1979) of each lake ranged from 3.89% d^{-1} in the main basin of Red Chalk Lake to 22.31% d^{-1} in Glen Lake with a mean of 7.1% d^{-1} (Table 92).

Using the three-year mean data ($n = 16$), community FR was strongly correlated ($p < 0.01$) with B_z and B_h; however, as community FR, B_z and B_h were all calculated

Table 91. Filtering rates (FR) of common herbivorous Crustacea in study lakes. Filtering rate calculated from average length of each taxon which was, in turn, calculated from average individual weight (w) of each taxon using indicated relationships.

Species	Weight (μg animal^{-1})	Length Code[1]	Length (mm)	Filtering Rate Code[2]	Filtering Rate (mL animal^{-1}d^{-1})	FR/W (mL^{-1} μg^{-1}d^{-1})
Bosmina longirostris	0.54	1	0.29	1	0.35	0.65
Ceriodaphnia sp.	0.50	2	0.44	2	1.71	3.42
Chydorus sp.	0.90	3	0.37	1	0.60	0.67
Daphnia ambigua	3.00	2	0.84	3	5.62	1.87
D. catawba	4.00	2	0.94	3	7.60	1.90
D. dubia	4.50	2	0.96	3	5.08	1.13
D. galeata mendotae	8.76	2	1.22	3	15.29	1.75
D. longiremis	5.00	2	1.00	3	8.97	1.79
D. pulex	8.09	2	1.18	4	9.89	1.22
D. retrocurva	3.14	2	0.85	3	5.81	1.85
D. rosea	8.00	2	1.18	5	24.09	3.01
Diaphanosoma sp.	1.89	2	0.71	6	6.48	3.43
Eubosmina longispina	0.54	1	0.29	1	0.35	0.65
E. tubicen	0.40	1	0.26	1	0.28	0.70
H. gibberum	10.90	4	1.19	7	14.68	1.35
nauplius	0.11	5	0.23	8	0.53	4.82
calanoid copepodid	0.87	6	0.47	8	1.03	1.18
Diaptomus minutus	2.24	6	0.69	8	1.47	0.66
D. oregonensis	3.55	6	0.84	8	1.77	0.50

[1] Length-weight relationships (from Sprules unpub. data)

1. $W = 266\ L^{3.13}$
2. $W = 50\ L^{2.84}$
3. $W = 297\ L^{3.48}$
4. $W = 70\ L^{2.56}$
5. $W = 42\ L^{2.48}$
6. $W = 55\ L^{2.46}$

[2] Filtering rates from P. Chow (unpub. studies)

1. $\log FR = 2.174 \log L + 0.713$
2. $\log FR = 3.523 \log L + 1.489$
3. $\log FR = 2.680 \log L + 0.953$
4. $\log FR = 1.998 \log L + 0.852$
5. $\log FR = 3.461 \log L + 1.133$
6. $\log FR = 2.421 \log L + 1.172$
7. $\log FR = 1.955 \log L + 1.019$
8. $\log FR = 0.94 \log L + 0.32$

from zooplankton densities this simply represents a self correlation.

Community FR was not correlated ($p > 0.10$) with 1977 to 1979 means of ice-free season values of Secchi transparency ($r = -0.08$), total chlorophyll *a* ($r = 0.17$), phytoplankton biomass ($r = 0.33$) or diatom corrected phytoplankton biomass ($r = 0.17$). However, as for B_z and B_h, community FR was correlated with TP ($r = 0.76$, $p < 0.01$). Given the strong B_z – TP correlation ($r = 0.80$, $p < 0.01$), this was not surprising.

The stepwise multiple linear regressions with Secchi depth, chlorophyll *a*, phytoplankton biomass and diatom corrected phytoplankton biomass (three-year averages of ice-free season and summer stratified season) as dependent variables were rerun with three-year average community FR ($n = 16$) added to the array of independent variables. Inclusion of FR did not effect any changes in resulting regression equations.

The eight regression equations contained 25 independent variables (as previously) and in only 1 of the 25 cases did community FR explain a significant ($p < 0.10$) portion of the residual variance in the dependent variable. Clearly, this could have been a chance occurrence. Hence, even though FR/W was not constant for all herbivores, explicit consideration of calculated filtering rates did not improve predictability of indicators of trophic state.

Table 92. Calculated community filtering rate (% d^{-1}) by herbivorous crustacean zooplankton in study lakes.

	Filtering Rate[1]			
Lake	1977	1978	1979	mean
Blue Chalk	5.05	6.02	8.45	6.51
Chub	4.50	4.12	5.46	4.69
Dickie	10.85	3.12	4.57	6.18
Harp	3.72	4.00	5.05	4.26
Jerry	2.70	4.17	7.61	4.83
Red Chalk Main	3.18	4.19	4.30	3.89
Red Chalk East	4.22	8.59	6.86	6.56
Basshaunt	–[2]	4.73	6.98	5.86
Bigwind	4.84	6.74	6.81	6.13
Buck	1.96	4.19	5.67	3.94
Crosson	5.59	9.85	9.10	8.18
Glen	–[2]	21.41	23.20	22.31
Gullfeather	4.20	14.42	13.91	10.84
Little Clear	8.17	10.14	8.69	9.00
Solitaire	3.59	4.98	7.20	5.26
Walker	2.36	4.05	8.74	5.05

[1] Filtering rate calculated by summing the products of average filtering rate of each species (mL L^{-1} d^{-1} $animal^{-1}$) with its associated average density (animals L^{-1}). Individual filtering rates in Table 89.

[2] no data

4. WATER CLARITY

Water clarity in lakes is governed by the amount (and size distribution) of particulate material, the amount and colour of dissolved organic matter and by the inherent absorption of light by pure water (Brezonik 1978). The particulate matter includes bacteria, pigment-containing phytoplankton, zooplankton, non-living autochthonous material (e.g., dead plant and animal matter), and organic and inorganic allochthonous materials (e.g., leaf litter particles, silt, clay, etc.). The use of water clarity as an indicator of trophic status is based on the assumption that transparency is principally determined by the quantity of pigment-containing phytoplankton. This belief has been supported by the observation that Secchi disc depth and chlorophyll *a* concentration are highly correlated in many lakes and sets of lakes (Edmondson 1972, Bachman and Jones 1974, Carlson 1977). In Ontario, Dillon and Rigler (1975) demonstrated that a hyperbolic relationship existed between mean chlorophyll *a* concentration and Secchi disc depth during summer stratification. The relationship can be written as:

$$SD_{SS} = 5.21 / (Chl\ a_{SS})^{0.41} \qquad (47)$$

where SD_{SS} is the mean Secchi disc depth (m) during summer stratification. However, it was observed that the predictive capability of this relationship for dystrophic lakes (those with high concentrations of dissolved organic matter) was poor. Attempts to construct a model to predict Secchi disc depth (during summer stratification and during the ice-free period) that considers dissolved organic matter in addition to particulate matter were therefore made.

Mean Secchi disc depth data collected for both the "A" and "B" lakes in each of the four years are summarized in Table 93. The four-year averages for the summer stratification and ice-free periods are summarized in Table 94. Variation between years was not great, with the average coefficient of variation being 15% and 13% for the two time periods. The range in mean Secchi disc depth was not large (2.7 m – 7.5 m). Models for prediction of the mean Secchi disc depth during the two time periods were derived using stepwise multiple regression. The independent variables utilized included all of those available for the chlorophyll *a* and phytoplankton biomass models (Table 75), plus the phytoplankton parameters themselves (chlorophyll *a*, total phytoplankton biomass, corrected phytoplankton biomass).

For both the ice-free and summer stratification periods, Secchi disc depth was most strongly correlated (Table 95) with mean annual dissolved organic carbon (DOC_{AN}). In each case, this variable explained ≃88% of the variance in Secchi disc depth. Secchi disc depth was also strongly correlated with all nitrogen fractions that included organic nitrogen (TON_{EP}, TN_{EP}, TKN_{EP}), with chlorophyll *a* (CHL_{SS}), total phosphorus (TP_{EP} and/or TP_{EM}) and water colour (COL_{IF}). DOC was very strongly correlated with water colour, each being a quantitative index of the degree of dystrophy of the lake. DOC was also strongly correlated with the nitrogen and phosphorus parameters, since the dissolved organic matter necessarily contains N and P as well as C.

Stepwise regression models, in both cases, selected TP (at spring overturn) after DOC (annual) as the second significant independent variable (Table 96). TP_{SO} has been previously used in many models to predict biomass of phytoplankton in the subsequent summer. However, none of the phytoplankton parameters themselves were selected as the second entry. This may be explained by

Table 93. Mean annual Secchi disc depth (m) for the summer stratification (SD_{SS}) and ice-free periods (SD_{IF}) for the years 1976 to 1979.

	SD_{SS}				SD_{IF}			
Lake	1976	1977	1978	1979	1976	1977	1978	1979
Blue Chalk	7.23	7.16	5.69	6.45	7.16	7.24	5.77	6.53
Chub	3.97	4.09	3.11	2.46	3.85	3.90	3.01	2.45
Dickie	3.12	3.18	3.03	2.28	2.96	2.98	2.96	2.30
Harp	4.41	4.33	3.40	3.56	4.46	4.32	3.37	3.54
Jerry	4.69	4.43	3.60	3.44	4.69	4.30	3.50	3.38
Red Chalk Main	6.76	6.50	5.50	6.21	6.65	6.46	5.42	5.96
Red Chalk East	–[1]	5.28	4.87	4.74	–[1]	5.22	4.80	4.68
Basshaunt	5.64	5.90	4.82	4.06	5.33	5.47	4.54	4.13
Bigwind	6.87	4.56	3.60	4.78	6.44	4.78	3.96	4.87
Buck	7.21	7.88	5.79	5.64	6.70	6.98	5.41	5.18
Crosson	3.80	4.53	3.62	3.22	3.74	4.10	3.37	3.00
Glen	4.48	–[1]	4.74	5.78	4.07	4.60	4.73	5.19
Gullfeather	2.80	3.22	2.38	2.46	2.73	2.90	2.18	2.33
Little Clear	5.91	6.27	4.85	4.74	5.62	5.46	4.92	4.74
Solitaire	7.94	8.30	7.00	6.82	7.57	8.42	7.05	7.55
Walker	5.47	6.70	5.35	4.98	5.24	6.62	5.05	5.29

[1] no data

the fact that TP may be a surrogate for all particulate material including live phytoplankton, living material other than phytoplankton, and non-living material. If this hypothesis were true, it would be expected that TN and TON would also be highly correlated (after DOC) with Secchi disc depth; this was, in fact, observed. Mean annual chlorophyll *a* concentration was selected as the third parameter in the ice-free period model, while maximum depth and mean depth were added to the model for the summer stratification period. Since the latter independent variables improved the predictive capabilities of the two models very little, models were constructed with only DOC_{AN} and TP_{SO} included (Table 97):

$$SD_{SS} = -1.03\ DOC_{AN} - 0.046\ TP_{SO} + 10.15 \qquad (48)$$

$$SD_{IF} = -1.03\ DOC_{AN} - 0.061\ TP_{SO} + 10.14 \qquad (49)$$

Table 94. Mean Secchi disc depth (m) in the summer stratification (SD_{SS}) and ice-free periods (SD_{IF}). Coefficient of variation (c.v.) between years expressed as %. Values are calculated as the means of 4 annual means for the years 1976 to 1980.

Lake	SD_{SS}	(c.v.)	SD_{IF}	(c.v.)
Blue Chalk	6.63	(11)	6.67	(10)
Chub	3.41	(23)	3.30	(21)
Dickie	2.90	(14)	2.80	(12)
Harp	3.92	(13)	3.92	(14)
Jerry	4.04	(15)	3.97	(16)
Red Chalk Main	6.24	(9)	6.12	(9)
Red Chalk East	4.96	(6)	4.90	(6)
Basshaunt	5.10	(16)	4.86	(13)
Bigwind	4.95	(28)	5.01	(21)
Buck	6.63	(17)	6.07	(15)
Crosson	3.79	(15)	3.55	(13)
Glen	5.00	(14)	4.65	(10)
Gullfeather	2.72	(14)	2.53	(13)
Little Clear	5.44	(14)	5.18	(8)
Solitaire	7.51	(10)	7.65	(7)
Walker	5.63	(13)	5.55	(13)
	mean	(15)	mean	(13)

Table 95. Correlation between mean Secchi disc depth during summer stratification (SS) and the ice-free periods (IF). Independent variables and their units are listed in Table 75.

SD_{SS}			SD_{IF}		
variable	r[1]	F[2]	variable	r[1]	F[2]
DOC_{AN}	–0.940	106.5	DOC_{AN}	–0.938	102.7
TON_{EP}	–0.835	32.3	TON_{EP}	–0.855	37.9
TN_{EP}	–0.835	32.3	TN_{EP}	–0.854	37.7
TKN_{EP}	–0.824	29.6	TKN_{EP}	–0.845	34.8
CHL_{SS}	–0.808	26.3	TP_{EP}	–0.818	28.4
TP_{EP}	–0.806	26.0	TP_{EM}	–0.805	25.8
COL_{IF}	–0.793	23.7	CHL_{SS}	–0.794	23.9

[1] correlation coefficient
[2] partial F-value

Table 96. Stepwise regression models for the prediction of Secchi disc depth in the summer stratification period (SD_{SS}) and the ice-free period (SD_{IF}). Independent parameters and their units are listed in Table 75.

	Step 1	Step 2	Step 3	Step 4	Constant	p	SE
SD_{SS}[1]	–0.99 DOC_{AN}	–0.075 TP_{SO}	–0.094 z_{MAX}	$0.20\bar{z}$	10.81	<0.001	0.344
SD_{IF}[2]	–1.26 DOC_{AN}	–0.065 TP_{SO}	–0.039 CHL_{AN}	–	10.27	<0.001	0.384

[1] $r^2 = 0.956$; $r_1^2 = 0.884$; $r_2^2 = 0.025$; $r_3^2 = 0.019$; $r_4^2 = 0.028$
[2] $r^2 = 0.941$; $r_1^2 = 0.880$; $r_2^2 = 0.042$; $r_3^2 = 0.019$

Table 97. Stepwise regression models truncated after second step for the prediction of average Secchi disc depth in the summer stratification period (SD_{SS}) and the ice-free period (SD_{IF}). Independent parameters and their units are listed in Table 75.

	Step 1	Step 2	Constant	p	SE
SD_{SS}[1]	$-1.03\ DOC_{AN}$	$-0.046\ TP_{SO}$	10.15	<0.001	0.454
SD_{IF}[2]	$-1.03\ DOC_{AN}$	$-0.061\ TP_{SO}$	10.14	<0.001	0.425

[1] $r^2 = 0.909$; $r_1^2 = 0.884$; $r_2^2 = 0.025$
[2] $r^2 = 0.922$; $r_1^2 = 0.880$; $r_2^2 = 0.042$

These models are recommended for predicting water clarity in lakes for which the range in independent variables falls within those observed in the study lakes. In cases where phytoplankton biomass or chlorophyll *a* concentration is much higher (>1 mg L^{-1}, or >5 $\mu g\ L^{-1}$, respectively), these models will overestimate water clarity.

E. AREAL HYPOLIMNETIC OXYGEN DEFICIT (AHOD)

The rate of loss of oxygen from the hypolimnion (or more properly, from the tropholytic zone – the zone where decomposition rate exceeds primary production rate) in lakes has been used for many years as a measure of trophic status (Hutchinson 1938). Although the loss of oxygen is a result of a combination of processes – decomposition of autochthonous matter that was produced in the euphotic zone then settled into the hypolimnion or tropholytic zone, decomposition of allochthonous organic matter (dissolved, colloidal and particulate), decomposition of particulate organic matter in the surficial sediments – the AHOD rate of a lake is an effective state variable dependent upon the overall trophic status of the lake.

Because hypolimnetic oxygen concentration determines, to a significant extent, the suitability of a lake for some fish populations, it would be useful to predict the AHOD, and in combination with lake morphometry, the oxygen concentration as a function of depth, in any lake. Only preliminary attempts to construct a model for the prediction of AHOD based on the Lakeshore Capacity Study lakes have been completed. A more detailed study of AHOD will be published separately.

In recent years, three models have been proposed for the prediction of AHOD. Charlton's (1980) model has been tested primarily for the Laurentian Great Lakes. Cornett and Rigler's (1979) and Welch and Perkins' (1979) models were developed using information collected for small lakes, more similar to those investigated in the Lakeshore Capacity Study. In light of this, the predictive capability of the Cornett and Welch models using AHOD's calculated for the "A" study lakes were compared.

Annual AHOD measured for each of the "A" lakes is shown in Table 98. Variation between lakes was small (less than two-fold), with variation between years as great or greater. Details of the oxygen regimes of the study lakes are provided by Reid et al. (1983a).

Table 98. Areal hypolimnetic oxygen depletion rate (AHOD; mg $O_2\ m^{-2}\ d^{-1}$) for the "A" lakes.

	AHOD				
Lake	1976	1977	1978	1979	mean
Blue Chalk	214	226	185	161	197
Chub	349	342	256	181	282
Dickie	305	259	261	271	274
Harp	429	501	343	240	378
Jerry	317	495	318	265	349
Red Chalk	307	481	316	186	323

The Welch model is:

$$\log AHOD = 1.58 + 0.37 \log (L_p \cdot \tau_w) \tag{50}$$

while the Cornett model can be expressed as:

$$AHOD = 0.5\,S - 277 + 150 \ln \bar{z}_H \tag{51}$$

where S is the TP retained in the lake each year (mg $m^{-2}\ yr^{-1}$), and $\bar{z}_H$ is the mean thickness of the hypolimnion.[1]

Total phosphorus input (Table 99) was calculated as the sum of the natural (measured) input plus, for those lakes with shoreline development, the inferred anthropogenic input (see Section III.B.10). The total phosphorus retained (S) was calculated as the difference between the total input and that lost by outflow (Section III.B.9). Comparison of the Welch and Cornett models indicated that although the predictions based on the Welch model were closer to the observed AHOD's than those based on the Cornett model, the correlation between predicted and observed values was much greater with the Cornett model. This fact suggested that re-calibration of the model coefficients would yield improved predictive capability. When this was done, the resulting Cornett-type model:

$$AHOD = 1.86\,S - 52 + 99.1 \ln \bar{z}_H \tag{52}$$

provided by far the closest agreement between measured and predicted AHOD's for the study lakes, with the mean square error reduced by 99% from the original Cornett model and 83% from the original Welch model.

The model represented by equation 52 must be tested on an independent data set before it can be considered to be a useful predictive model.

Table 99. Estimated TP load (sum of the measured natural load, Tables 53-58, and the estimated anthropogenic load – section III.B.10; in mg $m^{-2}\ yr^{-1}$) and the mean thickness of the hypolimnion (z_H, m).

	L_P					
Lake	1976-77	1977-78	1978-79	1979-80	mean	z_H
Blue Chalk	79.9	61.7	53.8	47.9	60.8	6.2
Chub	135.9	121.1	109.7	129.3	124.0	6.8
Dickie	197.1	176.0	166.1	178.8	179.5	1.5
Harp	143.7	115.4	125.1	112.2	124.1	10.0
Jerry	224.2	158.7	204.2	163.1	187.5	10.0
Red Chalk	101.6	98.8	89.3	87.2	94.2	11.2

[1] A third term, dependent upon hypolimnetic temperature, has been excluded for this analysis since its contribution to the model is very small.

F. STEPWISE PROCEDURE FOR CALCULATING THE EFFECT OF SHORELINE DEVELOPMENT ON THE TROPHIC STATUS OF LAKES

The conceptual model that formed the basis for the research carried out by the Trophic Status Component of the Lakeshore Capacity Study was illustrated in Figure 1. In sections III and IV, the outcome of this research, a series of mathematical relationships (models) describing the linkages in the conceptual model, was presented. This section provides a stepwise procedure describing use of the model for determining the effects of shoreline development on variables which describe the trophic status of a lake including the concentration of TP; algal abundance expressed as phytoplankton biomass and chlorophyll *a* concentration; water clarity as expressed by Secchi disc depth; and hypolimnetic oxygen conditions as expressed by the areal hypolimnetic oxygen deficit (AHOD).

The model can also be used to determine the capacity of a specific lake for development if an acceptable trophic status is specified.

The model described in this report was developed on the basis of data collected for a series of small, oligotrophic and mesotrophic lakes. Extrapolation of this model to lakes that are very different (e.g. extremely large, or eutrophic) is not warranted without further research.

Most of the information needed to utilize the model can be obtained from existing data bases including aerial photographs, topographic and geological maps, government reports, and other published literature. In some steps of the model, several options exist for performing the required calculations. Those options utilizing observed data rather than predicted information should be used wherever possible.

Step 1: Assemble basic information.

Obtain from existing data bases (or measure if necessary) lake surface area (A_o, m^2), lake volume (V, m^3) and the area of the subwatershed of each inlet to the lake (A_d, m^2). This information will be used in several steps of the model. Other basic information, required at certain steps or for certain calculation options, will be described as it is needed.

Step 2: Calculate anthropogenic TP input (JA).

Obtain information on the total number of units (n) (including any existing ones) to be situated on the shoreline of the lake (defined here as within 300 m of the lake or any inflowing stream to the lake). Additionally, obtain information on any proposed large-scale alterations in land use (e.g. clearcutting of forest, dredging or filling of lowland areas) which may affect the TP input from the terrestrial watershed.

Using the appropriate methodology outlined in Downing (1983), calculate the number of capita years yr^{-1} that will be spent at the lake (usage).

Assume a value of 0.8×10^6 mg capita^{-1} yr^{-1} as the anthropogenic TP supply to the sewage disposal facility. From information on the type of existing and proposed sewage systems, determine the retention capacity (R_s) of each system as follows:

$R_s = 0$ for conventional septic tank/tile field systems, leaching pits and aerobic systems. If there is evidence that the septic system is in good order and information on the soil properties is available, then calculate a value for R_s from Table 29.

$R_s = 1$ for holding tanks if waste disposal is outside of the watershed.

Calculate the anthropogenic TP supply to the lake (J_A, mg yr^{-1}) as follows:

$$J_A = 0.8 \times 10^6 . \text{usage} . (1 - R_s)$$

This calculation assumes that all anthropogenic TP reaches the lake the same year that it is supplied to the sewage disposal system (i.e. lag time = 0).

Step 3: Calculate TP input from precipitation (JPR).

Using the estimated TP deposition of 37.3 mg m^{-2} yr^{-1} and the lake area A_o (m^2), calculate the annual TP input (J_{PR}, mg yr^{-1}) to the lake from precipitation as follows:

$$J_{PR} = 37.3 . A_o$$

Step 4: Calculate natural TP input from terrestrial watershed (JN).

The method chosen to determine TP input from the watershed (J_N) is dependent upon the detail of information on watershed characteristics available. The following calculations must be performed for each inlet to the lake. If extensive data on the hydrological and geological characteristics of the watershed are available (Table 70), use the following relationship to predict TP export (E in mg m^{-2} yr^{-1}) from the watershed:

$$E = 0.38V27 + 0.012V2 - 0.33V8 - 0.29V28 - 23.85V43 + 1.43$$

If no hydrological information is available, calculate E as follows:

$$E = 0.46V27 - 0.27V8 - 0.28V28 + 0.053V29 + 5.99$$

If the geologic information is not sufficiently detailed to allow use of the above equation, TP export may be estimated knowing basic characteristics of land use and bedrock geology as outlined in Table 26.

Using the calculated values of E and the drainage basin area (A_d, m^2), calculate the TP input to the lake (J_N in mg yr^{-1}) from the n watersheds as follows:

$$J_N = \sum_{i=1}^{n} E_i . A_{di}$$

Take lakes upstream into account by starting the calculations at the headwater lake of the chain. The fraction of TP supplied to the lake that is retained by the lake is calculated as outlined in Step 6.

Step 5: Calculate hydrological characteristics of lake (Q, qs, τ_w).

The primary hydrological characteristic is lake outlet discharge (Q in m^3 yr^{-1}) and the method of estimation depends upon the amount and type of available information. If daily precipitation depth, daily meteorological data, and the physical and geological characteristics of the watershed are available, use the modified TVA model to calculate runoff. Instructions for the use of this model are

too lengthy to present in this summary. If this detailed information is not available and estimates of the long-term mean Q are adequate, use Pentland (1968) or the Hydrological Atlas of Canada (1978) to estimate runoff (r in m yr^{-1}). Annual lake outflow volume may be calculated as follows:

$Q = r \cdot A_d$

Calculate areal water loading rate (q_s in m yr^{-1}) as follows:

$q_s = Q/A_o$

and the theoretical water replenishment rate (τ_w in yr^{-1}) as follows:

$\tau_w = Q/V$

Step 6: Calculate retention of TP by lake (Rp).

Use existing information (or direct measurements) to determine the oxygen levels of the hypolimnion. If the hypolimnion experiences periods of anoxic conditions, the retention of TP by the lake is calculated as follows:

$R_p = 7.2/(7.2 + q_s)$

Otherwise, calculate the TP retention as follows:

$R_p = 12.4/(12.4 + q_s)$

As it is not currently possible to reliably predict the oxygen levels in the hypolimnion, measurements are necessary.

Step 7: Calculate the concentration of TP in the lake (TP_{IF}).

Using estimates of J_A, J_{PR} and J_N derived in steps 2-4, calculate the total load of TP to the lake (L in mg m^{-2} yr^{-1}) as follows:

$$L = \frac{J_A + J_{PR} + J_N}{A_o}$$

Calculate the mean ice-free concentration of TP in the lake as follows:

$[TP]_{IF} = L\,(1 - R_p)/0.956\, q_s$

If information is available on the concentration of TP in the lake at other time periods, calculate the mean ice-free period [TP] using the relations in Table 48.

Step 8: Calculate the response time of the lake (RT).

The response time of the lake (RT in years) gives an indication of the time required for the concentration of TP in the lake to respond to an altered TP supply. Response time can be expressed in terms of the half-life ($t_{1/2}$), the time required for the TP concentration in the lake to change half-way from its original steady-state level to its final steady-state level. It is suggested that 3 times the $t_{1/2}$ (or 87.5% of the time required to reach the final steady-state TP concentration) be used as an indicator of response time. Calculate response time as follows:

$$RT = 3t_{1/2} = 2.07/(1/\tau_w + v/\bar{z})$$

where $v = 12.4$ for lakes with oxic hypolimnia

and $v = 7.2$ for lakes with anoxic hypolimnia

Step 9: Calculate algal standing stock (chl a_{IF}, PB_{IF}, CPB_{IF}).

Three estimates of mean ice-free algal standing stock may be calculated (chl a_{IF} – chlorophyll *a*; PB_{IF} – phytoplankton biomass; CPB_{IF} – phytoplankton biomass excluding diatoms). The choice of equations used to estimate algal standing stock depends upon the type of available information. Where two equations are given, use the first one if the information is available. The independent variables used in the equations are listed in Table 75.

Calculate chlorophyll *a* concentrations (chl *a*, mg m^{-3}) using one of the two following equations:

$$(\text{chl } a)_{IF} = 0.83\, TP_{EP} + 0.12\,(N/P)_{EP} - 0.018\, TIN_{EP} - 0.0076\, TON_{IC} - 5.56$$

$$(\text{chl } a)_{IF} = 0.48\, TP_{EP} + 0.060\,(N/P)_{EP} - 3.14$$

If no data on N/P ratios in the epilimnion are available, use the following relation based on N/P ratios at fall overturn:

$$(N/P)_{EP} = 0.43\,(N/P)_{FO} + 19.4$$

Calculate phytoplankton biomass (PB_{IF} in mg m^{-3}) using the following equation:

$$PB_{IF} = 0.068\, TP_{EM} + 0.0044$$

Calculate phytoplankton biomass excluding diatoms (CPB_{IF}) using one of the following two equations:

$$CPB_{IF} = 0.25\, TP_{EP} + 0.039\,(N/P)_{EP} - 0.0053\, TN_{EP} + 0.080\, pH_{EP} - 2.10$$

$$CPB_{IF} = 0.098\, TP_{EP} + 0.012\,(N/P)_{EP} - 0.84$$

Step 10: Calculate water clarity (SDIF).

Based on the amount of data available, calculate the water clarity as expressed by Secchi depth (SD_{IF}) using one (use the first equation if the data is available) of the following equations:

$$SD_{IF} = -1.26\, DOC_{AN} - 0.065\, TP_{SO} - 0.39\,(\text{chl } a)_{AN} + 10.27$$

$$SD_{IF} = -1.03\, DOC_{AN} - 0.061\, TP_{SO} + 10.14$$

Measurements of DOC will have to be taken as no method currently exists for predicting DOC levels. If no data are available for TP_{SO}, it may be related to TP_{IF} using the following equation:

$$TP_{SO} = 1.18\, TP_{IF} - 1.91$$

Step 11: Calculate the areal hypolimnetic oxygen deficit (AHOD).

Calculate the areal hypolimnetic oxygen deficit (AHOD in mg 0_2 m^{-2} d^{-1}) from the following equation:

$$AHOD = 1.86\, S - 52 + 99.1 \ln \bar{z}_H$$

$\bar{z}_H$ (mean depth of the hypolimnion) may be estimated if the lake morphometry and the depth of the thermocline (D_T) are known. D_T may be calculated as follows (Ragotzkie 1978):

$$D_T = 4\sqrt{F}$$

where F = fetch of lake (km).

$S = L \cdot R_p$

As the equation predicting AHOD has not been tested on an independent data set as yet, it should be applied with caution.

If no data on $\bar{z}_H$ is available, calculate AHOD as follows:

$\log AHOD = 1.58 + 0.37 \log (L \cdot \tau_w)$

V. ADDITIONAL WORK

In addition to the components of the trophic status model described in the previous sections, preliminary work was carried out on two related subjects.

Zooplankton biomass in the study lakes was measured to investigate the possibility of improving nutrient-phytoplankton relationships (see Section IV.D.3). However, the potential for predicting zooplankton biomass was also explored since the capability of predicting zooplankton biomass may be an important intermediate step in predicting fish biomass. This is discussed in the subsequent sub-section.

In the trophic status modelling that was undertaken in this study, total phosphorus was the only form of phosphorus considered. Reasons for the emphasis on this fraction of phosphorus have been discussed by Dillon and Reid (1981). However, the fraction of TP that is biologically available (BAP) is generally unknown. If appropriate methods of measurement of BAP existed, better trophic status models could almost certainly be produced.

The first steps towards development of a suitable method of measuring BAP were made as part of this study. A bioassay based on phosphorus uptake rate measurements was successfully developed. The method is described in detail by Dillon and Reid (1981) and Mierle (1983). A preliminary investigation of the availability of the phosphorus in some of the major inputs to the study lakes demonstrated that the TP input via atmospheric deposition was totally available to biological organisms, while only a fraction (10-60%) of that in several of the streams was available. These results indicate that the TP from different sources may not be of equal importance in determining the trophic status of lakes. These studies will be described in detail in a separate report.

A. EMPIRICAL PHOSPHORUS – ZOOPLANKTON BIOMASS RELATIONSHIPS

In recent years limnologists have developed a large number of empirical regression models which can be used to predict phytoplankton biomass (PB) or chlorophyll *a* (chlor *a*). Fisheries biologists similarly have developed empirical relationships for use in predicting fish community yield or biomass from such independent variables as total dissolved solids (Rawson 1951, 1960) mean depth (Rawson 1952, Prepas 1983), the ratio of total dissolved solids to mean depth (Ryder et al. 1974), phytoplankton productivity or biomass (Oglesby 1977, Biro and Vörös 1982), macrobenthic biomass (Matuszek 1978) and, finally, TP levels (Hanson and Leggett 1982).

Since the pioneering work of Rawson (1953) and Northcote and Larkin (1956) there has been little effort directed at developing such models for zooplankton biomass. Given the intermediate tropho-dynamic position of zooplankton between phytoplankton and fish, such models may be of interest. Surveys of existing literature from lakes in various regions have indicated that over large gradients in nutrient status good correlations may exist between phytoplankton biomass or productivity and zooplankton biomass or productivity (Brylinsky and Mann 1973, Mackarewicz and Likens 1979, McCauley and Kalff 1981). However no relationships have been constructed (since Northcote and Larkin 1956) using data collected with that purpose in mind from a single geographic region, for example, for Canadian Shield Lakes. Further, the usefulness of nutrients, particularly TP, as predictors of zooplankton biomass has only recently been investigated (Hanson and Peters 1984).

Earlier in this report (Section IV.D), it was shown that TP was a good predictor of PB in the study lakes even though the range in TP levels was small. In this section, the usefulness of readily obtainable morphometric, chemical and phytoplanktonic information for predicting average total crustacean zooplankton biomass (B_Z) in the LCS lakes is assessed. As zooplankton may be important in the diet of Ontario fish (Martin 1952, Keast 1973), it is anticipated that such predictions would be of interest to fisheries biologists.

Zooplankton were enumerated in whole-lake volume-weight composite samples collected roughly on a weekly and monthly basis from the "A" and "B" lakes, respectively, as described by Scheider et al. (1983). Biomass in each sample (mg m^{-3} dry weight) was calculated from the enumerated samples, using average species dry weights. The reported values of B_Z are time-weighted arithmetic means of biomass for the ice-free seasons of 1977, 1978 and 1979. Three-year means are used in an attempt to reduce uncertainty associated with low total numbers of samples in the "B" lakes (Table 88) and to eliminate the effects of lags in population responses which may confound shorter-term data.

Among the "A" lakes, B_Z varied by less than two-fold from 37.1 mg m^{-3} in the main basin of Red Chalk Lake to 58.2 mg m^{-3} in the small eastern basin. B_Z ranged from 33 to 104 mg m^{-3} among the "B" lakes (Table 88). These values are at the low end of the range of values typically observed in oligotrophic Shield lakes (Schindler and Novén 1971).

Product moment correlation coefficients of B_Z with 16 limnological variables are given in Table 100. Correla-

tions were poor ($p > 0.10$) for Secchi depth, chlor *a* and PB, for the chemical parameters, pH, colour, DOC and TIN/TP and for two morphometric parameters, A_o and V. Significant positive correlations were observed for TP, TON, TIN and TN, and significant negative correlations with TN/TP, $\bar{z}$ and z_{max}. While not indicated in the table, correlations with annual and summer stratified averages were of similar sign and magnitude as those with ice-free season averages.

Table 100. Correlation coefficients (r) between B_z and a variety of limnological variables. All chemical and biological data were three year means. Single and double asterisks indicate $P < 0.05$ and $P < 0.01$, respectively.

Parameter	r
TP	0.80**
TON	0.52*
TIN	0.70**
TN	0.75**
TN/TP	–0.54*
TIN/TP	–0.05
Secchi	–0.30
Chlor *a*	0.28
PB	0.38
pH	0.29
Colour	0.40
DOC	0.30
A_o	–0.26
$\bar{z}$	–0.57*
z_m	–0.65**
V	–0.42*

There are three possible reasons why the correlation of B_z with nitrogen and particularly with phosphorus levels may have been stronger than that with expressions of phytoplankton standing stock. First, herbivorous zooplankton feed on bacteria and tripton (non-living suspended material) in addition to phytoplankton. Gliwicz (1969) in fact estimated that in oligotrophic lakes in Poland, phytoplankton form a maximum of 50% of the diet of planktonic herbivores. Measured TP is probably a better measure of total grazable material in the study lakes than is PB or chlor *a*.

Secondly, the impact of herbivores on phytoplankton is negative at times, but as recyclers of essential nutrients zooplankton may also have a net positive impact on phytoplankton at other times (Lehman 1980, Redfield 1980). The poor correlations of B_z with PB may, therefore, be a reflection of temporal changes in the kind of phytoplankton-zooplankton interaction. Finally, the fraction of PB which is actually grazable may be overestimated by BP and may vary between lakes, as zooplankton may graze selectively upon the available phytoplankton species (McNaught et al. 1980).

All variables listed in Table 1 were used as the independent variable matrix in a stepwise multiple linear regression analysis using B_z as the dependent variable. With the F ratio to add or delete variables set at 95%, the resulting model had TP as the single independent variable:

$$B_z = 14.32 + 4.46\,\mathrm{TP} \quad (r^2 = 0.63, F = 24.1, P < 0.01)$$

Given the successful application of TP as a predictor of community standing stocks at other trophic levels, it is not surprising that it is the best univariate predictor of zooplankton biomass in the study lakes.

McCauley and Kalff (1981) assembled phytoplankton biomass and zooplankton biomass data from 17 lakes in various parts of the world in an effort to construct a PB-B_z relationship. Over their wide range in PB (0.4 to 50 mg L^{-1} wet weight), they found a good relationship existed between the parameters (both in $\mu g\ L^{-1}$ wet weight) as below:

$$\log_{10} B_z = 0.719 \log_{10} \mathrm{PB} + 1.015 \qquad (r^2 = 0.72)$$

Because B_z and PB were weakly correlated in the study lakes (Table 100) it was not anticipated that this relationship could be used to predict B_z in the lakes. To test this, the data were converted to wet weights assuming a dry to wet weight ratio of 0.12, as did McCauley and Kalff, and predicted B_z using measured PB values. As expected, predicted and measured B_z values were very poorly correlated ($r = 0.39$, $p > 0.10$, $n = 16$), and the slope of an equation of predicted vs. observed B_z was only 0.2, much less than the ideal of 1.0. This indicates that the McCauley and Kalff relationship is not useful for predicting B_z in nutrient-poor Shield lakes.

McCauley and Kalff did not examine TP-B_z relationships to see if, as in the study lakes, these may have been stronger than PB-B_z relationships. The TP data were not available (McCauley, personal communication). An examination of two other data sets indicates that over a wider range in nutrient status than found in the study lakes, TP may indeed be a better predictor of B_z than PB. For seven lakes in the Kawartha region of Ontario with TP levels ranging from 16 to 55 mg m^{-3}, TP and B_z were strongly correlated ($r = 0.92$, $p < 0.01$), but the correlation of PB with B_z was poor ($r = 0.46$, $p > 0.10$, Anon. 1976). Similarly, while Mills and Shiavone (1982) report a significant relationship between chlor *a* and B_z for 13 lakes in New York State ($r = 0.75$), correlation of B_z with TP levels calculated from presented data are stronger ($r = 0.84$).

VI. SUMMARY

The Lakeshore Capacity Study was initiated to provide quantitative models that would allow evaluation of the effects of shoreline development on lakes and their watersheds. This report summarizes the research carried out to estimate the effects of shoreline development on the trophic status of lakes. The results of these studies have been integrated with those of the other components to provide a sound technical basis for management of shoreline development.

The conceptual model linking shoreline development, morphometry, the hydrologic and TP budgets and trophic status of a lake was originally outlined by Dillon and Rigler (1975). The trophic status of a lake was described by several state variables including phytoplankton biomass, chlorophyll *a* (a surrogate measure of phytoplankton biomass), water clarity and the loss rate of oxygen from the profundal waters.

A critical untested assumption in the conceptual model was that shoreline development resulted in the input of TP to lakes via the sewage disposal systems. Furthermore, many linkages in the conceptual model were not sufficiently well developed to allow accurate predictions of trophic status with an acceptable degree of confidence. Further model development and testing were therefore necessary.

Two approaches were used in this study to address the question of whether shoreline development provided TP to lakes. In the first approach, the natural TP and hydrologic budgets were measured for six lakes ("A" lakes) exhibiting a large range in the potential significance of shoreline development to the TP budgets. Existing trophic status models were calibrated using the two undeveloped (no anthropogenic TP input) lakes of the set, then applied to the four developed lakes. Differences between measured TP concentrations in the developed lakes and TP concentrations predicted from the calibrated models were assumed to be attributable to anthropogenic TP inputs which, in turn, were quantitatively related to the extent of shoreline development.

In the second approach, the trophic status of nine selected lakes ("B" lakes) was compared before and after shoreline development. However, as few of these changes occurred over the course of the Lakeshore Capacity Study, this approach is not discussed at length in this report.

During the course of the study, relationships were developed, tested and refined to link individual components of the conceptual model. The models developed include those for prediction of the important components of the hydrology of lakes and watersheds, the yield or export of TP from the terrestrial watershed, the retention of TP in the sediments of the lake, the concentration of TP in the lake, the phytoplankton biomass and chlorophyll *a* levels, the water clarity (as measured by Secchi disc), the zooplankton biomass, and the depletion rate of oxygen from the profundal waters. Because several of these relationships depended on the geology and geochemistry of the watersheds, an additional set of nine terrestrial watersheds ("export" watersheds) were chosen to extend the range in geological characteristics observed in the "A" and "B" lake watersheds.

Details of the methods employed in carrying out the studies are described in Scheider et al. (1983a). The study lakes are located in the County of Haliburton and the District Municipality of Muskoka in Ontario. Information on climate, lake morphometry, vegetation and geology are summarized in Scheider et al. (1983b), Nicolls et al. (1983), and Jeffries and Snyder (1983) respectively. All lakes exhibited pH, conductivity and major ion and nutrient concentrations typical for those observed in dilute, nutrient-poor Precambrian lakes. Average (1976-1980) concentrations of TP ranged from 5.7 mg m^{-3} to 12.8 mg m^{-3} for the "A" lakes and 6.4 mg m^{-3} to 20.7 mg m^{-3} for the "B" lakes. Using the trophic status classification scheme of Vollenweider and Kerekes (1980), all lakes but one could be classified as oligotrophic. A summary of the major findings regarding the individual components of the hydrologic and TP budgets of the "A" lakes follows.

Hydrologic budgets were calculated for each lake for four annual (June-May) periods and for the entire four-year period. In all cases except one, the budgets balanced to within ±10%, thereby giving confidence in the individual supply and loss terms and in the use of data for construction of chemical mass balances and the calculation of water replenishment times for the lakes. A bias to negative balances (inputs < losses) was observed.

The major ion balances for the conservative ions Ca^{2+}, Mg^{2+}, Na^+, K^+, and Cl^- generally supported the contention that the hydrological budgets were accurately measured. The ion balances showed deficits in input, supporting the findings of the hydrologic budgets, and indicating a small systematic error in the mass balances.

The range in atmospheric deposition of TP was 21.8 mg m^{-2} yr^{-1} to 58.4 mg m^{-2} yr^{-1} with a mean (1976-1980) of 37.3 mg m^{-2} yr^{-1}, indicating the need for long-term monitoring to obtain reliable estimates of deposition. A seasonal pattern in the deposition of TP was observed, peak values occurring in the summer months. Deposition rates of TP at all but one location in the study area were not significantly different, although significant differences were observed between collectors located on a

lake and those located under the forest canopy adjacent to the lake.

The mean (1976-1980) TP export (yield) for 30 watersheds ranged from 2.0 mg m^{-2} yr^{-1} to 28.5 mg m^{-2} yr^{-1}. The grand mean TP export for metamorphic watersheds was 8.2 mg m^{-2} yr^{-1}, somewhat higher than the mean (5.5 mg m^{-2} yr^{-1}) based on previous studies. Year-to-year variability in yield was not significant compared to site-to-site variability, although the monthly values of TP yield ranged over two orders of magnitude within each watershed with maximum values occurring during the spring snowmelt.

Based on a literature review, 0.8 kg TP $capita^{-1}$ yr^{-1} was calculated as the average TP supply to sewage disposal systems in the area. Estimates of TP retention in the sewage disposal system were made for types of systems in use around the study lakes. Retention of TP in septic tank and tile field systems was assumed to be zero unless information on the condition of the system and soil type was available. A correction factor for TP retention in the soil may then be applied. Information on human usage and sewage disposal type obtained from the Land Use Component (Downing, pers. comm.) allowed estimates of potential anthropogenic TP input to the study lakes ranging from 0 kg yr^{-1} to 82.8 kg yr^{-1}.

The loss of TP to the lake sediments was calculated in three ways: from the measured TP mass balances (undeveloped lakes only); from sediment trap data; and from information on the concentration of TP in sediments and the sediment accumulation rate. A comparison of the latter two methods with the first method showed that sediment traps overestimated the TP sedimentation rate by 12 to 17 times, whereas good agreement was found between the mass balance and sediment chronology methods.

Minor components of the TP budget were investigated. For Harp Lake, net TP loss by fish harvest was estimated as 2% of the total TP supply, TP loss by emerging insects was estimated as being negligible and the TP supply by forest litterfall was estimated as 3% of the total TP input.

Based on the TP mass balance information, the relative importance of the various sources of TP to the lakes were assessed. The inflowing streams were of highly variable importance, providing from 0.4% to 69% of the natural TP input to the lakes. Precipitation input of TP was the single most important source of TP in four of the six "A" lakes, contributing 20%-74% of the total natural supply (1976-1980 mean values). Contributions of TP from the ungauged watershed areas were relatively small, averaging 15% of the total natural supply of TP to the lakes (range 8%-23%). The loss of TP to the sediments was always greater than the loss to the lake outlet, even if the potential anthropogenic inputs were not considered. Finally, the potential contributions of the anthropogenic sources of TP were calculated as 55%, 46%, 19%, 7%, 1% and 0% of the total TP supply for Harp, Dickie, Blue Chalk, Chub, Red Chalk and Jerry Lakes respectively. The year-to-year variability in the individual components of the TP budgets was great, indicating that measurement of TP mass balances for several years was necessary to develop and validate trophic status models.

The impacts of shoreline development on communities of phytoplankton, zooplankton and macrophytes were assessed on the "A" and "B" lakes by examining the correlations between parameters describing the state of the biological communities and indices of lakeshore development or by comparing the values of the parameters in sets of lakes of extremely different intensities of use. Of the 18 correlations run for phytoplankton, only one was significant and the development index explained only 28% of the variance in the parameter. Comparing phytoplankton community parameters for the three most developed and the four undeveloped lakes showed that there was no significant difference between the two sets of lakes. No detectable change was found in the composition, standing stock or nutritional status of macrophytes in eight lakes covering the full range of human use. Finally, comparison of ten zooplankton community parameters (1977-1979) in the three most developed and four undeveloped lakes showed no significant differences between lake sets.

A summary of the relationships developed, refined and tested to link individual components of the conceptual model are given below.

Existing models predicting the four principal hydrologic components (precipitation, runoff, evapotranspiration and evaporation) from readily obtainable hydrometeorological data were refined and tested on the "A" lakes and "export" watersheds. Precipitation at a given site could be predicted using data from nearby, existing meteorological stations. Of the five techniques investigated to predict runoff, the preferred method was the modified Tennessee Valley Authority (TVA) model because it allowed prediction of streamflow on a fine time scale (daily) for any given set of climatic conditions. After calibration on the "A" lake watersheds, the modified TVA model was tested on the "export" watersheds and gave good predictions of daily flow in all years except 1977-78. Both basin evapotranspiration and lake evaporation could be predicted based on the climatological models of Morton et al. (1980). For Harp Lake, estimates of annual lake evaporation using Morton's method agreed to within 2%-8% of those made using the more laborious energy balance method. Evapotranspiration rates for Harp and Jerry Lake watersheds estimated using Morton's method agreed to within 2% of those estimated using the water balance method over the 1976-1980 period.

Two models were developed to predict annual yield of TP from terrestrial watersheds. In the first model, 54% of the variance in the TP yield of 30 watersheds was related to geologic, land use and hydrologic characteristics of the watershed. In the second model, 46% of the variance in the TP yield was explained by geologic and land use characteristics alone.

The relationship given in Dillon et al. (1977) successfully predicted the annual retention of TP from the areal water load when tested against estimates derived from the TP mass balances in Jerry and Red Chalk (Main Basin) Lakes. The model was recalibrated for anoxic lakes (on Red Chalk Lake, East Basin) resulting in an adjustment of the settling velocity of TP from 12.4 m yr^{-1} to 7.2 m yr^{-1}.

An existing model for prediction of concentration of TP in lakes at spring turnover was refined to allow prediction of TP in any season and to incorporate the relation between the concentration of TP in the lake and that in the lake outlet. In lakes with no anthropogenic input of TP, the predicted and observed mean (1976-1980) concentrations of TP during the ice-free season agreed very closely (9.3 $\mu g\ L^{-1}$ vs. 9.3 $\mu g\ L^{-1}$ in Jerry Lake and 5.5 $\mu g\ L^{-1}$ vs. 5.6 $\mu g\ L^{-1}$ in the Main Basin of Red Chalk Lake), giving confidence in the use of the model in lakes with anthropogenic inputs of TP. The incorporation of thermal stratification in the model did not improve its predictive capability.

Using the revised TP retention and concentration models, the predicted and observed concentrations of TP were compared for the four "A" lakes with shoreline development and the fraction of the potential anthropogenic TP input entering the lakes was calculated. For Chub, Blue Chalk and Dickie Lakes, the predicted and observed TP levels agreed to within 1 $\mu g\ L^{-1}$ if it was assumed that 100% of the anthropogenic TP supply entered the lakes. However, for Harp Lake, only 20% of the anthropogenic TP supply must enter the lake for the observed and predicted TP levels to agree. This difference is likely due to differences in the TP retention capability of the soils surrounding the lake but this was not quantified.

Models were developed on the "A" and "B" lake data to predict the phytoplankton biomass, 'corrected' phytoplankton biomass (excluding diatoms) and the concentration of chlorophyll *a* (a correlate of phytoplankton biomass) during the ice-free season from epilimnetic concentrations of algal nutrients (1976-1980 mean values). Chlorophyll *a* and 'corrected' phytoplankton biomass could both be predicted from the concentration of TP and the ratio of the concentrations of TN to TP. The phytoplankton biomass was related to the concentration of total organic N. The inclusion of total zooplankton biomass or herbivorous zooplankton biomass in the relations did not improve their predictive ability.

Water clarity (as measured by Secchi depth) of the "A" and "B" lakes during the ice-free season was controlled by the dissolved organic carbon content (or colour) of the lakes. Chlorophyll *a* concentration and phytoplankton biomass were not strongly correlated with water clarity.

The models of Cornett and Rigler (1979) and Welch and Perkins (1979) for prediction of areal hypolimnetic oxygen rates were tested. The Cornett-Rigler model, when recalibrated with data collected in this study, provided the closest agreement between predicted and observed depletion rates.

Total crustacean zooplankton biomass and herbivorous zooplankton biomass of the "A" and "B" lakes (1977-1979 mean values) could be predicted from the whole-lake concentrations of TP during the summer stratified period. These predictions may be useful in future efforts to model fish biomass.

Finally, a step-by-step methodology for predicting the effects of shoreline development on the trophic status of lakes is provided.

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APPENDIX

Tables A1 – A6.

Annual water balances for six study lakes.

Tables A7 – A12.

Major ion mass balances for six study lakes.

Tables A13 – A18.

Total phosphorus mass balances for six study lakes.

Table A1. Annual water balances for Harp Lake for 1976-77, 1977-78, 1978-79, 1979-80 and 4 year balance for 1976-80. Individual supply and loss terms in $10^3 m^3 yr^{-1}$ and as % total supply or % total loss.

	1976-1977		1977-1978		1978-1979		1979-1980		1976-1980	
	$10^3 m^3 yr^{-1}$	%	$10^3 m^3 yr^{-1}$	%	$10^3 m^3 yr^{-1}$	%	$10^3 m^3 yr^{-1}$	%	$10^3 m^3$	%
Supply terms										
Precipitation	605	24.7	606	22.2	763	20.7	630	17.2	2,604	20.8
Inlet 3	105	4.3	114	4.2	144	3.9	144	3.9	507	4.1
3A	58.8	2.4	132	4.8	141	3.8	119	3.3	451	3.6
4	458	18.7	571	20.9	911	24.7	771	21.1	2,711	21.7
5	864	35.3	851	31.2	1,130	30.6	1,420	38.8	4,265	34.1
6	43.2	1.8	56.5	2.1	82	2.2	69.7	1.9	251	2.0
6A	52.0	2.1	63.1	2.3	83	2.3	86.6	2.4	285	2.3
Ungauged	259	10.6	333	12.2	435	11.8	423	11.6	1,450	11.6
Total	2,445	99.9	2,727	99.9	3,689	100.0	3,663	100.2	12,524	100.0
Loss terms										
Outlet	1,770	78.4	2,530	87.1	3,550	88.7	3,400	88.9	11,250	86.6
Evaporation	488	21.6	374	12.9	452	11.3	424	11.1	1,738	13.4
Total	2,258	100.0	2,904	100.0	4,002	100.0	3,824	100.0	12,988	100.0
lake storage	0	–	98.3	–	16.1	–	–68.2	–	46.2	–
Balance[1]	187	8.0	–275	–9.8	–329	–8.6	–92.8	–2.5	–510	–3.6

[1] Balance = supply – loss ± Δ storage

$$\% \text{ balance} = (\text{balance}/\frac{\text{supply} + \text{loss}}{2})\ 100$$

Table A2. Annual water balances for Jerry Lake for 1976-77, 1977-78, 1978-79, 1979-80 and 4 year balance for 1976-80. Individual supply and loss terms in $10^3 m^3 yr^{-1}$ and as % total supply or % total loss.

	1976-1977		1977-1978		1978-1979		1979-1980		1976-1980	
	$10^3 m^3 yr^{-1}$	%	$10^3 m^3 yr^{-1}$	%	$10^3 m^3 yr^{-1}$	%	$10^3 m^3 yr^{-1}$	%	$10^3 m^3$	%
Supply terms										
Precipitation	453	12.1	454	11.5	571	10.6	471	9.2	1,949	10.7
Inlet 1	22.9	0.6	30.5	0.8	38.5	0.7	37	0.7	129	0.7
3	2,660	71.0	2,740	69.6	3,800	70.6	3,660	71.2	12,860	70.6
4	122	3.3	141	3.6	201	3.7	209	4.1	673	3.7
Ungauged	491	13.1	570	14.5	771	14.3	763	14.8	2,595	14.3
Total	3,749	100.1	3,936	100.0	5,382	99.9	5,140	100.0	18,206	100.0
Loss terms										
Outlet	3,590	90.3	3,840	93.1	5,520	94.1	4,460	93.2	17,410	92.8
Evaporation	386	9.7	286	6.9	348	5.9	328	6.8	1,348	7.2
Total	3,976	100.0	4,126	100.0	5,868	100.0	4,788	100.0	18,758	100.0
lake storage	46.6		29.1		–15.0		–71.1		–57.0	
Balance[1]	–274	–7.1	–219	–5.4	–471	–8.4	+423	8.5	–495	–2.7

[1] Balance = supply – loss ± Δ storage

$$\% \text{ balance} = \left(\text{balance}/\frac{\text{supply} + \text{loss}}{2}\right) 100$$

Table A3. Annual water balances for Chub Lake for 1976-77, 1977-78, 1978-79, 1979-80 and 4 year balance for 1976-80. Individual supply and loss terms in $10^3 m^3 yr^{-1}$ and as % total supply or % total loss.

	1976-1977		1977-1978		1978-1979		1979-1980		1976-1980	
	$10^3 m^3 yr^{-1}$	%	$10^3 m^3 yr^{-1}$	%	$10^3 m^3 yr^{-1}$	%	$10^3 m^3 yr^{-1}$	%	$10^3 m^3$	%
Supply terms										
Precipitation	265	22.2	263	17.9	435	25.0	380	21.0	1,343	21.6
Inlet 1	206	17.3	216	14.7	287	16.5	260	14.4	969	15.6
2	453	38.0	649	44.1	638	36.7	763	42.2	2,503	40.3
Ungauged	269	22.6	343	23.3	378	21.8	405	22.4	1,395	22.5
Total	1,193	100.1	1,471	100.0	1,738	100.0	1,808	100.0	6,210	100.0
Loss terms										
Outlet	863	76.9	1,360	88.5	1,640	88.4	1,800	89.3	5,663	86.7
Evaporation	260	23.1	176	11.5	216	11.6	215	10.7	867	13.3
Total	1,123	100.0	1,536	100.0	1,856	100.0	2,015	100.0	6,530	100.0
lake storage	–15.1		24.2		19.0		–30.0		–1.9	
Balance[1]	85.1	7.3	–89.2	–5.9	–137	–7.6	–177	–9.3	–318	–5.0

[1] Balance = supply – loss ± Δ storage

$$\% \text{ balance} = \left(\text{balance}/\frac{\text{supply} + \text{loss}}{2}\right) 100$$

Table A4. Annual water balances for Dickie Lake for 1976-77, 1977-78, 1978-79, 1979-80 and 4 year balance for 1976-80. Individual supply and loss terms in $10^3 m^3 yr^{-1}$ and as % total supply or % total loss.

	1976-1977		1977-1978		1978-1979		1979-1980		1976-1980	
	$10^3 m^3 yr^{-1}$	%	$10^3 m^3 yr^{-1}$	%	$10^3 m^3 yr^{-1}$	%	$10^3 m^3 yr^{-1}$	%	$10^3 m^3$	%
Supply terms										
Precipitation	791	32.1	751	28.6	1,258	35.6	1,100	29.5	3,900	31.6
Inlet 5	175	7.1	159	6.1	190	5.4	226	6.1	750	6.1
6	98.7	4.0	118	4.5	151	4.3	145	3.9	513	4.2
8	187	7.6	230	8.8	218	6.2	400	10.7	1,035	8.4
10	353	14.3	358	13.6	457	12.9	495	13.3	1,663	13.5
11	325	13.2	424	16.2	548	15.5	548	14.7	1,845	15.0
Ungauged	533	21.6	584	22.3	709	20.1	811	21.8	2,637	21.4
Total	2,463	99.9	2,624	100.1	3,531	100.0	3,725	100.0	12,340	100.2
Loss terms										
Outlet	1,900	72.6	2,200	78.9	2,690	79.8	2,910	82.8	9,700	78.9
Evaporation	716	27.4	580	21.1	681	20.2	605	17.2	2,582	21.1
Total	2,616	100.0	2,780	100.0	3,371	100.0	3,515	100.0	12,280	100.0
lake storage	0		105		18.6		–144		–20.5	
Balance[1]	–153	–6.0	–261	–9.7	141	4.1	354	9.8	81.5	0.7

[1] Balance = supply – loss ± Δ storage

% balance = $(\text{balance}/\frac{\text{supply + loss}}{2})\ 100$

Table A5. Annual water balances for Blue Chalk Lake for 1976-77, 1977-78, 1978-79, 1979-80 and 4 year balance for 1976-80. Individual supply and loss terms in $10^3 m^3 yr^{-1}$ and as % total supply or % total loss.

	1976-1977		1977-1978		1978-1979		1979-1980		1976-1980	
	$10^3 m^3 yr^{-1}$	%	$10^3 m^3 yr^{-1}$	%	$10^3 m^3 yr^{-1}$	%	$10^3 m^3 yr^{-1}$	%	$10^3 m^3$	%
Supply terms										
Precipitation	407	49.1	404	45.3	667	52.9	583	48.9	2,061	49.4
Inlet 1	85.6	10.3	53.2	6.0	62.6	5.0	108	9.1	309	7.4
Ungauged	336	40.5	435	48.7	530	42.1	500	42.0	1,801	43.2
Total	829	99.9	892	100.0	1,260	100.0	1,191	100.0	4,171	100.0
Loss terms										
Outlet	644	63.2	713	71.2	962	74.1	1,030	77.6	3,349	72.1
Evaporation	375	36.8	288	28.8	337	25.9	297	22.4	1,297	27.9
Total	1,019	100.0	1,001	100.0	1,299	100.0	1,327	100.0	4,646	100.0
lake storage	–113		77.1		21.2		–16.8		–31.1	
Balance[1]	–77	–8.3	–186	–19.7	–60.2	–4.7	–119	–9.5	–444	–10.1

[1] Balance = supply – loss ± Δ storage

% balance = $(\text{balance}/\frac{\text{supply + loss}}{2})\ 100$

Table A6. Annual water balances for Red Chalk Lake for 1976-77, 1977-78, 1978-79, 1979-80 and 4 year balance for 1976-80. Individual supply and loss terms in $10^3m^3yr^{-1}$ and as % total supply or % total loss.

	1976-1977		1977-1978		1978-1979		1979-1980		1976-1980	
	$10^3m^3yr^{-1}$	%	$10^3m^3yr^{-1}$	%	$10^3m^3yr^{-1}$	%	$10^3m^3yr^{-1}$	%	10^3m^3	%
Supply terms										
Precipitation	468	17.5	465	14.9	768	18.9	671	17.0	2,372	17.2
Inlet 1	648	24.3	695	22.3	872	21.5	885	22.4	3,100	22.4
2	94	3.5	120	3.9	175	4.3	149	3.8	538	3.9
3	368	13.9	507	16.3	550	13.6	533	13.5	1,958	14.2
4	163	6.1	250	8.0	281	6.9	265	6.7	959	7.0
Blue Chalk outlet	644	24.1	713	22.9	962	23.7	1,030	26.0	3,349	24.3
Ungauged	286	10.7	369	11.8	451	11.1	425	10.7	1,531	11.1
Total	2,671	100.1	3,119	100.1	4,059	100.0	3,958	100.1	13,807	100.1
Loss terms										
Outlet	2,490	85.8	2,940	90.1	4,040	91.6	3,970	91.3	13,440	90.0
Evaporation RC main	320	11.0	249	7.6	283	6.4	291	6.7	1,143	7.7
Evaporation RC east	93.3	3.2	75.1	2.3	88.8	2.0	85.4	2.0	343	2.3
Total	2,903	100.0	3,264	100.0	4,412	100.0	4,346	100.0	14,925	100.0
lake storage	0		64.9		18.2		–12.0		71.1	
Balance[1]	–232	–8.3	–210	–6.6	–371	–8.8	–376	–9.1	–1,189	–8.3

[1] Balance = supply – loss ± Δ storage

$$\% \text{ balance} = \left(\text{balance} / \frac{\text{supply} + \text{loss}}{2}\right) 100$$

Table A7. Ca^{2+}, Mg^{2+}, Na^{+}, K^{+} and Cl^{-} mass balances for Harp Lake for 4 12-month periods and the total 4-year period. Input includes that from all gauged inflows, from the ungauged area and from atmospheric deposition, output consists of loss via outflow, and Δ storage represents the change in mass in the lake as a result of change in lake water level. The difference (Δ) is expressed as the surplus or deficit relative to the output, and the % is calculated relative to the output.

	1976-77	1977-78	1978-79	1979-80	1976-80
Ca^{2+}					
Input	6,960	8,500	10,600	10,300	36,360
Δ Storage	0	+316	+55	−228	+143
Output	5,720	9,100	12,500	11,000	38,320
Δ	+1,240	−916	−1,955	−472	−2,103
%	+21.7	−10.1	−15.6	−4.3	−5.5
Mg^{2+}					
Input	2,140	2,410	2,820	2,620	9,990
Δ Storage	0	+97	+15	−60	+52
Output	1,790	2,610	3,400	2,880	10,680
Δ	+350	−297	−595	−200	−742
%	+16.4	−11.4	−17.5	−6.9	−7.0
Na^{+}					
Input	2,140	2,690	3,550	2,910	11,290
Δ Storage	0	+97	+17	−70	+44
Output	1,920	2,750	3,860	3,650	12,180
Δ	+220	−157	−327	−670	−934
%	+11.5	−5.7	−8.5	−18.4	−7.7
K^{+}					
Input	1,170	1,130	1,160	1,500	4,960
Δ Storage	0	+55	+9	−38	+26
Output	1,050	1,510	1,900	1,990	6,450
Δ	+120	−435	−749	−452	−1,516
%	+11.4	−28.8	−39.4	−22.7	−23.5
Cl^{-}					
Input	1,780	2,140	2,530	2,620	9,070
Δ Storage	0	+71	+12	−55	+28
Output	1,350	2,050	3,020	3,050	9,470
Δ	+430	+19	−502	−375	−428
%	+24.2	+0.9	−16.6	−12.3	−4.5

Table A8. Ca^{2+}, Mg^{2+}, Na^{+}, K^{+} and Cl^{-} mass balances for Jerry Lake for 4 12-month periods and the total 4-year period. Input includes that from all gauged inflows, from the ungauged area and from atmospheric deposition, output consists of loss via outflow, and Δ storage represents the change in mass in the lake as a result of change in lake water level. The difference (Δ) is expressed as the surplus or deficit relative to the output, and the % is calculated relative to the output.

	1976-77	1977-78	1978-79	1979-80	1976-80
Ca^{2+}					
Input	12,500	15,200	19,100	17,500	64,300
Δ Storage	+168	+103	−57	−252	−38
Output	12,200	14,700	21,500	15,600	64,000
Δ	+132	+397	−2,343	+2,152	+338
%	+1.1	+2.7	−10.9	+13.8	+0.5
Mg^{2+}					
Input	4,330	4,740	5,480	4,560	19,110
Δ Storage	+56	+34	−18	−71	+1
Output	4,080	4,720	6,100	4,220	19,120
Δ	+194	−14	−602	+411	−11
%	+4.8	−0.3	−9.9	+9.7	−0.1
Na^{+}					
Input	3,400	4,130	5,300	4,450	17,280
Δ Storage	+40	+27	−16	−64	−13
Output	3,220	4,150	7,040	4,280	18,690
Δ	+140	−47	−1,724	+234	−1,397
%	+4.4	−1.1	−24.5	+5.5	−7.5
K^{+}					
Input	1,880	2,110	2,420	2,610	9,020
Δ Storage	+27	+16	−8	−37	−2
Output	2,010	2,170	2,650	2,400	9,230
Δ	−157	−76	−222	+247	−208
%	−7.8	−3.5	−8.4	+10.3	−2.3
Cl^{-}					
Input	2,600	2,570	3,400	2,780	11,350
Δ Storage	+24	+16	−8	−41	−9
Output	1,990	2,180	3,060	2,360	9,590
Δ	+586	+374	+348	+461	+1,769
%	+29.5	+17.2	+11.4	+19.5	+18.5

Table A9. Ca^{2+}, Mg^{2+}, Na^{+}, K^{+} and Cl^{-} mass balances for Dickie Lake for 4 12-month periods and the total 4-year period. Input includes that from all gauged inflows, from the ungauged area and from atmospheric deposition, output consists of loss via outflow, and Δ storage represents the change in mass in the lake as a result of change in lake water level. The difference (Δ) is expressed as the surplus or deficit relative to the output, and the % is calculated relative to the output.

	1976-77	1977-78	1978-79	1979-80	1976-80
Ca^{2+}					
Input	5,460	5,840	7,070	7,560	25,930
Δ Storage	0	+289	+52	−395	−54
Output	4,980	6,140	7,200	8,040	26,360
Δ	+480	−589	−182	−85	−376
%	+9.6	−9.6	−2.5	−1.1	−1.4
Mg^{2+}					
Input	1,300	1,340	1,550	1,630	5,820
Δ Storage	0	+72	+12	−84	0
Output	1,220	1,490	1,660	1,590	5,960
Δ	80	−222	−122	+124	−140
%	+6.6	−14.9	−7.4	+7.8	−2.3
Na^{+}					
Input	1,630	2,190	2,310	2,100	8,230
Δ Storage	0	+89	+17	−123	−17
Output	1,560	1,940	2,470	2,530	8,500
Δ	+70	+161	−177	−307	−253
%	+4.5	+8.3	−7.2	−12.1	−3.0
K^{+}					
Input	999	903	943	1,130	3,975
Δ Storage	0	+48	+8	−60	−4
Output	831	995	1,080	1,110	4,016
Δ	+168	−140	−145	+80	−37
%	+20.2	−14.1	−13.4	+7.2	−0.9
Cl^{-}					
Input	2,380	2,690	2,850	3,040	10,960
Δ Storage	0	+92	+16	−125	−17
Output	1,690	2,120	2,420	2,600	8,830
Δ	+690	+478	+414	+565	+2,147
%	+40.8	+22.6	+17.1	+21.7	+24.3

Table A10. Ca^{2+}, Mg^{2+}, Na^{+}, K^{+} and Cl^{-} mass balances for Chub Lake for 4 12-month periods and the total 4-year period. Input includes that from all gauged inflows, from the ungauged area and from atmospheric deposition, output consists of loss via outflow, and Δ storage represents the change in mass in the lake as a result of change in lake water level. The difference (Δ) is expressed as the surplus or deficit relative to the output, and the % is calculated relative to the output.

	1976-77	1977-78	1978-79	1979-80	1976-80
Ca^{2+}					
Input	3,610	4,250	4,470	4,600	16,930
Δ Storage	−43	+72	+58	−87	0
Output	2,580	4,190	4,760	4,870	16,400
Δ	+1,073	−12	−348	−183	+530
%	+41.6	−0.3	−7.3	−3.8	+3.2
Mg^{2+}					
Input	927	1,140	1,030	1,040	4,137
Δ Storage	−11	+18	+14	−19	+2
Output	657	1,060	1,120	1,100	3,937
Δ	+281	+62	−104	−41	+198
%	+42.8	+5.9	−9.3	−3.7	+5.0
Na^{+}					
Input	904	1,330	1,380	1,250	4,864
Δ Storage	−11	+19	+15	−22	+1
Output	634	1,110	1,580	1,290	4,614
Δ	+281	+201	−215	−18	+249
%	+44.3	+18.1	−13.6	−1.4	+5.4
K^{+}					
Input	721	635	608	691	2,655
Δ Storage	−7	+11	+8	−12	0
Output	392	632	646	734	2,404
Δ	+336	−8	−46	−31	+251
%	+85.7	−1.3	−7.1	−4.2	+10.4
Cl^{-}					
Input	1,170	1,240	1,150	1,230	4,790
Δ Storage	−9	+15	+11	−18	−1
Output	538	955	987	1,030	2,510
Δ	+641	+270	+152	+218	+1,281
%	+119.1	+28.3	+15.4	+21.2	+36.5

Table A11. Ca^{2+}, Mg^{2+}, Na^{+}, K^{+} and Cl^{-} mass balances for Blue Chalk Lake for 4 12-month periods and the total 4-year period. Input includes that from all gauged inflows, from the ungauged area and from atmospheric deposition, output consists of loss via outflow, and Δ storage represents the change in mass in the lake as a result of change in lake water level. The difference (Δ) is expressed as the surplus or deficit relative to the output, and the % is calculated relative to the output.

	1976-77	1977-78	1978-79	1979-80	1976-80
Ca^{2+}					
Input	1,680	1,840	2,010	1,970	7,500
Δ Storage	–319	+224	+64	–48	–79
Output	1,920	2,230	3,120	2,870	10,140
Δ	+79	–614	–1,174	–852	–2,561
%	+4.1	–27.5	–37.6	–29.7	–25.3
Mg^{2+}					
Input	386	434	433	411	1,664
Δ Storage	–91	+61	+16	–11	–25
Output	513	567	743	677	2,500
Δ	–36	–194	–326	–255	–811
%	–7.0	–34.2	–43.9	–37.7	–32.4
Na^{+}					
Input	481	636	648	535	2,300
Δ Storage	–95	+64	+18	–14	–27
Output	547	616	967	890	3,020
Δ	+29	–44	–337	–341	–693
%	+5.3	–7.1	–34.9	–38.3	–23.0
K^{+}					
Input	292	271	227	287	1,077
Δ Storage	–43	+30	+8	–6	–11
Output	241	300	439	422	1,402
Δ	+94	–59	–220	–129	–314
%	+39.0	–19.7	–50.1	–30.6	–22.4
Cl^{-}					
Input	421	436	439	416	1,712
Δ Storage	–48	+36	+8	–8	–12
Output	426	331	474	477	1,708
Δ	+43	+69	–43	–53	+16
%	+10.1	+20.9	–9.1	–11.1	+0.9

Table A12. Ca^{2+}, Mg^{2+}, Na^{+}, K^{+} and Cl^{-} mass balances for Red Chalk Lake for 4 12-month periods and the total 4-year period. Input includes that from all gauged inflows, from the ungauged area and from atmospheric deposition, output consists of loss via outflow, and Δ storage represents the change in mass in the lake as a result of change in lake water level. The difference (Δ) is expressed as the surplus or deficit relative to the output, and the % is calculated relative to the output.

	1976-77	1977-78	1978-79	1979-80	1976-80
Ca^{2+}					
Input	7,120	9,030	10,500	9,280	35,930
Δ Storage	0	+190	+56	–35	+211
Output	6,810	9,100	12,400	10,500	38,810
Δ	+310	–260	–1,956	–1,185	–3,091
%	+4.6	–2.9	–15.8	–11.3	–8.0
Mg^{2+}					
Input	1,860	2,340	2,490	2,130	8,820
Δ Storage	0	+55	+15	–8	+62
Output	1,950	2,720	3,280	2,690	10,640
Δ	–90	–435	–805	–552	–1,882
%	–4.6	–16.0	–24.5	–20.5	–17.7
Na^{+}					
Input	1,940	2,620	3,120	2,650	10,330
Δ Storage	0	+52	+15	–10	+57
Output	1,780	2,350	3,060	3,210	10,400
Δ	+160	+218	+45	–550	–127
%	+9.0	+9.3	+1.5	–17.1	–1.2
K^{+}					
Input	1,060	1,280	1,240	1,320	4,900
Δ Storage	0	+29	+8	–5	+32
Output	1,030	1,390	1,640	1,680	5,740
Δ	+30	–139	–408	–355	–872
%	+2.9	–10.0	–24.9	–21.1	–15.2
Cl^{-}					
Input	1,620	1,730	1,950	1,810	7,110
Δ Storage	0	+30	+9	–6	+33
Output	1,190	1,390	1,870	1,730	6,180
Δ	+430	+310	+71	+86	+897
%	+36.1	+22.3	+3.8	+5.0	+14.5

Table A13. Summary of total phosphorus mass balance for Harp Lake. Input from the ungauged area was calculated on the basis of the measured total phosphorus gross export (yield per unit area) of the 6 gauged subwatersheds. Results are in kg yr^{-1}, figures in brackets are per cent of total measured input contributed by source.

Source	1976-77	1977-78	1978-79	1979-80	1976-80
Inflow 3	3.29 (4.1)	2.47 (4.0)	3.26 (4.8)	2.84 (4.8)	11.9 (4.4)
Inflow 3A	0.47 (0.6)	0.88 (1.4)	0.73 (1.1)	0.74 (1.2)	2.82 (1.0)
Inflow 4	7.44 (9.2)	10.4 (16.8)	11.9 (17.5)	10.9 (18.4)	40.6 (15.0)
Inflow 5	23.9 (29.5)	17.0 (27.5)	23.9 (35.1)	21.2 (35.7)	86.0 (31.8)
Inflow 6	0.75 (1.0)	0.81 (1.3)	0.99 (1.5)	0.88 (1.5)	3.43 (1.3)
Inflow 6A	0.68 (0.8)	0.73 (1.2)	0.84 (1.2)	1.84 (3.1)	4.09 (1.5)
Ungauged	5.39 (6.7)	5.07 (8.2)	6.21 (9.1)	6.34 (10.7)	23.0 (8.5)
Total terrestrial	41.9 (51.7)	37.4 (60.5)	47.8 (70.3)	44.7 (75.3)	171.8 (63.6)
Atmosphere	39.1 (48.3)	24.4 (39.5)	20.2 (29.7)	14.6 (24.6)	98.3 (36.4)
Total measured	81.0 (100.0)	61.8 (100.0)	68.0 (100.0)	59.4 (99.9)	270.2 (100.0)
Development[1]	79.8	81.2	82.5	82.5	326.0
Outflow	14.3	16.9	27.2	26.4	84.8

[1] Estimated in Table 32.

Table A14. Summary of total phosphorus mass balance for Dickie Lake. Input from the ungauged area was calculated on the basis of the measured total phosphorus gross export (yield per unit area) of the 5 gauged subwatersheds. Results are in kg yr^{-1}, figures in brackets are per cent of total measured input contributed by source.

Source	1976-77	1977-78	1978-79	1979-80	1976-80
Inflow 5	6.45 (6.0)	4.85 (5.6)	3.85 (5.1)	6.09 (7.0)	21.2 (5.9)
Inflow 6	3.78 (3.5)	5.89 (6.8)	7.75 (10.2)	11.5 (13.1)	28.9 (8.1)
Inflow 8	3.33 (3.1)	3.93 (4.5)	2.32 (3.1)	4.97 (5.7)	14.6 (4.1)
Inflow 10	15.3 (14.1)	11.1 (12.8)	8.79 (11.6)	10.1 (11.5)	45.3 (12.6)
Inflow 11	7.05 (6.5)	8.87 (10.2)	7.29 (9.6)	9.29 (10.6)	32.5 (9.1)
Ungauged	18.1 (16.7)	18.2 (21.0)	17.5 (23.1)	25.3 (28.9)	79.1 (22.1)
Total terrestrial	54.0 (49.8)	52.8 (60.8)	47.5 (62.8)	67.3 (76.9)	221.6 (61.8)
Atmosphere	54.4 (50.2)	34.0 (39.2)	28.2 (37.3)	20.3 (23.2)	136.9 (38.2)
Total measured	108.4 (100.0)	86.8 (100.0)	75.7 (100.1)	87.5 (100.1)	358.4 (100.0)
Development[1]	75.3	77.2	79.1	79.1	310.7
Outflow	22.8	27.3	33.2	31.7	115.0

[1] Estimated in Table 32.

Table A15. Summary of total phosphorus mass balance for Chub Lake. Input from the ungauged area was calculated on the basis of the measured total phosphorus gross export (yield per unit area) of the 2 gauged subwatersheds. Results are in kg yr^{-1}, figures in brackets are per cent of total measured input contributed by source.

Source	1976-77	1977-78	1978-79	1979-80	1976-80
Inflow 1	1.55 (3.8)	2.24 (6.2)	2.30 (7.1)	2.11 (5.4)	8.20 (5.5)
Inflow 2	14.9 (36.2)	15.7 (43.3)	14.2 (43.8)	21.2 (54.6)	66.0 (44.4)
Ungauged	6.02 (14.6)	6.67 (18.4)	6.20 (19.1)	8.51 (21.9)	27.4 (18.4)
Total terrestrial	22.5 (54.6)	24.6 (67.8)	22.7 (70.0)	31.8 (81.9)	101.6 (68.3)
Atmosphere	18.8 (45.6)	11.7 (32.2)	9.74 (30.0)	7.02 (18.1)	47.3 (31.8)
Total measured	41.2 (100.2)	36.3 (100.0)	32.5 (100.0)	38.8 (100.0)	148.8 (100.1)
Development[1]	2.57	2.70	2.82	2.82	10.9
Outflow	8.43	13.9	15.0	17.9	55.2

[1] Estimated in Table 32.

Table A16. Summary of total phosphorus mass balance for Blue Chalk Lake. Input from the ungauged area was calculated on the basis of the measured total phosphorus gross export (yield per unit area) of the 5 gauged subwatersheds of Blue Chalk and Red Chalk Lakes. Results are in kg yr^{-1}, figures in brackets are per cent of total measured input contributed by source.

Source	1976-77	1977-78	1978-79	1979-80	1976-80
Inflow 1	0.43 (1.3)	0.47 (1.9)	0.29 (1.4)	0.99 (5.5)	2.18 (2.2)
Ungauged	4.56 (13.5)	6.30 (25.4)	5.70 (27.3)	6.26 (34.7)	22.8 (23.4)
Total terrestrial	4.99 (14.8)	6.77 (27.3)	5.99 (28.7)	7.25 (40.2)	25.0 (25.6)
Atmosphere	28.8 (85.2)	18.0 (72.6)	14.9 (71.3)	10.8 (59.8)	72.5 (74.4)
Total measured	33.8 (100.0)	24.8 (99.9)	20.9 (100.0)	18.0 (100.0)	97.5 (100.0)
Development[1]	5.66	5.66	5.66	5.66	22.6
Outflow	3.49	4.86	5.57	6.51	20.4

[1] Estimated in Table 32.

Table A17. Summary of total phosphorus mass balance for Red Chalk Lake. Input from the ungauged area was calculated on the basis of the measured total phosphorus gross export (yield per unit area) of the 5 gauged subwatersheds of Blue Chalk and Red Chalk Lakes. Results are in kg yr^{-1}, figures in brackets are per cent of total measured input contributed by source.

Source	1976-77	1977-78	1978-79	1979-80	1976-80
Inflow 1	6.23 (10.8)	9.32 (16.6)	9.58 (18.9)	12.5 (25.2)	37.6 (17.5)
Inflow 2	1.31 (2.3)	1.69 (3.0)	2.01 (4.0)	1.67 (3.4)	6.68 (3.1)
Inflow 3	4.31 (7.5)	7.83 (13.9)	6.44 (12.7)	6.20 (12.5)	24.8 (11.6)
Inflow 4	5.32 (9.2)	6.41 (11.4)	5.19 (10.2)	4.93 (9.9)	21.9 (10.2)
Blue Chalk outflow	3.49 (6.0)	4.86 (8.6)	5.57 (11.0)	6.51 (13.1)	20.4 (9.5)
Ungauged	3.88 (6.7)	5.35 (9.5)	4.84 (9.5)	5.32 (10.7)	19.4 (9.0)
Total terrestrial	24.5 (42.5)	35.5 (63.1)	33.6 (66.2)	37.1 (74.9)	130.7 (61.0)
Atmosphere	33.2 (57.4)	20.7 (36.8)	17.2 (33.9)	12.4 (25.0)	83.5 (38.9)
Total measured	57.8 (99.9)	56.2 (99.9)	50.8 (100.1)	49.6 (99.9)	214.4 (99.9)
Development[1]	0.60	0.60	0.60	0.60	2.40
Outflow	12.2	20.8	24.6	26.3	83.9

[1] Estimated in Table 32.

Table A18. Summary of total phosphorus mass balance for Jerry Lake. Input from the ungauged area was calculated on the basis of the measured total phosphorus gross export (yield per unit area) of the 3 gauged subwatersheds. Results are in kg yr^{-1}, figures in brackets are per cent of total measured input contributed by source.

Source	1976-77	1977-78	1978-79	1979-80	1976-80
Inflow 1	0.29 (0.3)	0.29 (0.4)	0.38 (0.4)	0.38 (0.5)	1.34 (0.4)
Inflow 3	71.4 (63.6)	52.5 (66.0)	74.6 (72.9)	59.3 (72.6)	257.8 (68.6)
Inflow 4	1.92 (1.7)	1.23 (1.6)	1.93 (1.9)	1.92 (2.4)	7.00 (1.9)
Ungauged	9.42 (8.4)	7.19 (9.0)	10.2 (10.0)	9.13 (11.2)	35.9 (9.6)
Total terrestrial	83.0 (73.9)	61.2 (77.0)	87.1 (85.1)	70.7 (86.5)	302.0 (80.4)
Atmosphere	29.3 (26.1)	18.3 (23.0)	15.2 (14.9)	10.9 (13.3)	73.7 (19.6)
Total measured	112.3 (100.0)	79.5 (100.0)	102.3 (100.0)	81.7 (99.8)	375.8 (100.0)
Development[1]	0.0	0.0	0.0	0.0	0.0
Outflow	29.9	27.7	45.9	47.4	150.9

[1] Estimated in Table 32.

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